

A MICROMETEOROLOGY STUDY OF STOCK WATERING PONDS, RANGELANDS,
AND WOODLANDS IN THE FLINT HILLS OF KANSAS

by

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B.S., Texas Tech University, 2000
M.S., Texas Tech University, 2002

AN ABSTRACT OF A DISSERTATION

submitted in partial fulfillment of the requirements for the degree

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College of Agriculture

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Abstract

Land management practices such as burning and grazing may affect evapotranspiration (ET) and water balance of the tall grass prairie in the Flint Hills of eastern Kansas. Experiments were designed to estimate or measure the water balance of a stock-watering pond, and compare energy balance parameters and ET between grazed and ungrazed prairies. The hydrology of the native tallgrass prairie also was compared with mature stand of eastern red cedar (*Juniperus virginiana*), a site that was formerly prairie but converted to woodland when prescribed burning was discontinued. Data were collected to encompass the seasonal and yearly changes in weather variability. A host of micrometeorological sensors were used to measure surface atmosphere exchange and water losses, including: eddy covariance towers on prairie and woodland sites, specialized throughfall and stem flow equipment at the woodland site, and an instrumentation raft at the stock-watering pond. Results of the stock pond study showed that, on average, evaporation accounted for 64% of the water loss, followed by seepage at 31%, cattle use at 3% and transpiration at 2%. Comparisons of grazed and ungrazed areas showed that grazing caused only small, 3 to 6 %, reductions in seasonal ET compared with ungrazed pastures despite large differences in vegetative cover. In the woodland study, the 50-yr-old cedar canopy intercepted 54% of the precipitation received, thus decreasing the amount of precipitation reaching the soil. Evapotranspiration from woodland and prairie sites were similar, but net carbon exchange was greater on the prairie. Thus, the apparent water use efficiency during the summer months was 3 times greater on the prairie. Net radiation at the woodland site was 100 W m⁻² higher compared with the prairie. This caused an increase in the woodland sensible heat flux and midday Bowen

ratios, yet woodland latent heat flux and ET was similar to the prairie during the study, factors that could affect regional climate. Land management decisions regarding grazing, prescribed burning, and stock pond design will impact the watershed hydrology and productivity of the tallgrass prairie.

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Table of Contents

List of Figures	xi
List of Tables	xiv
List of Tables	xiv
Acknowledgements	xv
Dedication.....	xvi
CHAPTER 1 - Introduction.....	1
Introduction.....	1
Objectives	6
References	7
CHAPTER 2 - Water Balance of a Stock-Watering Pond in the Flint Hills of Kansas.....	10
Introduction.....	10
Methods	14
Site Description	14
Water Balance Measurements and Calculations	15
Modeling Pond Evaporation	19
Results	20
Weather and Evaporative Demand.....	20
Pond Depth Changes and Runoff.....	21
Seepage	22
Evaporation	23
Cattle Consumption and Transpiration.....	24

Partitioning Water Losses during the Grazing Season	25
Evaporation Modeling	25
DISCUSSION.....	27
IMPLICATIONS	28
Reference List	43
CHAPTER 3 - The Effect of Grazing on Evapotranspiration and Net Carbon Exchange in	
Tallgrass Prairie	47
Introduction.....	47
Materials and Methods	50
Eddy Covariance and Meteorological Measurements	51
Results	53
Biomass, leaf area index, and climate	53
Evapotranspiration.....	54
Net Carbon Exchange and NCE:ET	55
Discussion.....	56
Implications	59
Reference List	71
CHAPTER 4 - Conversion of Tallgrass Prairie to <i>Juniperus virginiana</i> : Effects on Rainfall	
Interception, Evapotranspiration, and Net Carbon Exchange	76
Introduction.....	76
Methods	80
Eddy Covariance and Meteorological Measurements	81
Measurements of Rainfall Interception	82

Soil and Plant Measurements	83
Results	84
Soil Water Data	84
Rainfall interception Study	84
Prairie vs. Woodland Evapotranspiration	86
Net carbon exchange	86
Prairie vs. Woodland Energy Balance	88
Discussion.....	90
Implications	92
Acknowledgements	93
References	109
CHAPTER 5 - Conclusion	116

List of Figures

Figure 2-1. Comparison of precipitation during 2005 and 2006 to the 30-yr (1971-2001) average for Manhattan, Kansas.	30
Figure 2-2. Depth of the pond at the deepest point as measured by the depth recorder in 2005 and 2006.	31
Figure 2-3. Total daily pond water losses for the entire study as measured by the floating depth recorder. The graph represents the change in pond depth, or the combined losses from E, T, C, and S over time.	32
Figure 2-4. Change in depth and evaporation from the pond during a 20-day seepage test between December 27, 2005 and January 15, 2006. Seepage rate was calculated as the difference between total depth change and cumulative evaporation over time. The apparent seepage rate for the test was 2.6 mm d^{-1}	33
Figure 2-5. Fluctuations in the apparent seepage rate for the stock pond over the entire study period. Since seepage is influenced by hydraulic head pressure at the soil liner, seepage rates fluctuate as pond depth changes.	34
Figure 2-6. Evaporation from the stock-watering pond during the grazing period of the study. Evaporation was measured using instrumentation on the meteorological raft floating on the center of the pond.	35
Figure 2-7. Cattle consumption and transpiration from the pond during the 2006 grazing season. Data are expressed in terms of pond depth.	36

Figure 2-8. Comparison of monthly evaporation measured from pond instrumentation to estimated evaporation derived from the Priestley-Taylor and Penman models using local weather data from the Konza Prairie Biological Station.	37
Figure 2-9. The correlation between measured monthly evaporation using the bulk transfer equation to the Priestley – Taylor (---) and to the Penman (—) evaporation models. Modeled values are from weather data obtained from the Konza Biological Station, and measured values are from the actual evaporation measured from raft instrumentation on the pond.	38
Figure 3-1. Seasonal aboveground biomass for the ungrazed and grazed treatments (error bars represent the standard error of the samples).	61
Figure 3-2. Seasonal trends in green leaf area index (LAI) among years compared between the ungrazed and grazed sites (error bars represent the standard error of the sample).	62
Figure 3-3. Monthly precipitation and reference crop evapotranspiration (ET _o) during the study.	63
Figure 3-4. Monthly evapotranspiration as measured by the eddy covariance towers for the ungrazed and grazed treatments.	64
Figure 3-5. Comparison of measured monthly ET for the ungrazed (—) and grazed (---) sites to the FAO-56 reference crop ET _o calculated from weather data.	65
Figure 3-6. Ratio of measured ET to ET _o on a monthly basis from the ungrazed (a) and grazed (b) sites to monthly total ET _o calculated from weather station data.....	66
Figure 3-7. Monthly total net carbon exchange (NCE) compared between the ungrazed and grazed treatments. Negative values represent a gain of carbon to the system (i.e., downward next flux).....	67

Figure 3-8. The ratio of NCE to ET compared between the ungrazed and grazed treatments.	
Negative numbers indicate the system was gaining carbon (net downward carbon flux).....	68
Figure 4-1. Photograph of a throughfall collector (area: 0.30 x 2.44 m) positioned below the woodland canopy.....	94
Figure 4-2. Photograph of a stemflow collector attached to a tree (total length: 735mm).	95
Figure 4-3. Volumetric soil water content (0 to 10cm) at the prairie and woodland sites.	96
Figure 4-4. Measurements of throughfall, stemflow, and precipitation at the woodland site. Also included is interception as calculated from equation 1.....	97
Figure 4-5. The comparison of throughfall and precipitation at the woodland site.....	98
Figure 4-6. Percent interception compared to precipitation rate at the woodland site.....	99
Figure 4-7. The monthly evapotranspiration for the Prairie and Woodland sites in 2004.	100
Figure 4-8. The monthly NCE from the Prairie and Woodland sites in 2004. The negative values represent a gain of carbon to the system (i.e., net downward flux).....	101
Figure 4-9. The ratio of NCE to ET compared between the prairie and woodland sites. Negative numbers represent a gain of carbon to the system.	102
Figure 4-10. The midday net radiation compared between the prairie and woodland sites during August, 2004. Data are averages calculated from 11:00 to 15:00 h local standard time...	103
Figure 4-11. Midday sensible heat flux (A), and latent heat flux (B), compared between the prairie and woodland sites during August, 2004. Data are averages calculated from 11:00 to 15:00 h local standard time.	104
Figure 4-12. Midday Bowen ratio (H/LE) compared between the prairie and woodland sites during August, 2004. Data are averages calculated from 11:00 to 15:00 h local standard time.	105

List of Tables

Table 2-1. Factors affecting the pond water balance (see Eq. 1).....	39
Table 2-2. Comparison of the measured pond inflow to modeled runoff using the NRCS curve number method. *.....	40
Table 2-3. The total monthly losses of water from the stock-watering pond during the 2005 and 2006 grazing period.	41
Table 2-4. Monthly weather parameters recorded from the weather station located at the Konza Prairie Station and the stock-watering pond. Shown in the table are the average air temperature, vapor pressure deficit (VPD), and wind speed for three months of the pond study.....	42
Table 3-1. The stocking date, density, initial and ending weights, and the average gain for the cattle put to pasture on intensive early stock grazed pasture.	69
Table 3-2. The yearly total evapotranspiration (ET), net carbon exchange (NCE), and the ratio of NCE to ET compared between the ungrazed and grazed sites. Negative numbers represent a sink (or gain) of carbon to the system.	70
Table 4-1. Recorded throughfall by individual sample location. Also included is the ratio of throughfall and precipitation.	106
Table 4-2. Recorded stemflow by individual collector.....	107

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Dedication

I would like to dedicate this dissertation to my family. Whether it was my mom and dad's visits to Kansas, or the laughter of my nieces and nephews on my trips home, my family has brightened my life. Their constant love and support has given me much needed encouragement and inspiration throughout all of my work in graduate school.

CHAPTER 1 - Introduction

Introduction

The Flint Hills region contains one of the largest unbroken spreads of native tallgrass in North America. Though much of the fertile Great Plains has been converted into crop production, the Flint Hills region remains blanketed in native tallgrass. The tallgrass prairie evolved under the influence of fire and grazing Axelrod (1985), and spring burning and grazing are two of the management practices that are used in sustaining the tallgrass prairie. The tallgrass prairie serves as a building block to the stocker cattle industry. Many pastures are burned each spring to remove the accumulated grass residue and mulch to promote vigorous regrowth of vegetation Adams and Anderson (1978). Burning allows more rapid soil warming and greatly stimulates growth and results in increased cattle gains Anderson et al. (1970). Furthermore, burning helps prevent woodland encroachment by killing young trees and shrubs. Burning is often completed in April, several weeks before stocker steers fill the prairies. Without prescribed burning, the prairie may eventually convert to a closed stand of eastern red cedar (*Juniperus virginiana*). The loss of prairie to woody encroachment has an immediate economic impact by reducing the number and productivity of acreage that can be used for grazing. Furthermore, these forested areas will have radically different energy and water balances than the prairie. Changes in the energy and water balance could alter regional scale hydrology and weather patterns. Therefore, patchy woodland establishment could ultimately affect the productivity of adjacent grasslands. Grazing strategies can also modify the surface energy balance and the water balance of the landscape. In general, the tallgrass prairie ecosystem is water limited, with potential

evapotranspiration far exceeding actual evapotranspiration Frank and Inouye (1994). Thus, any management factor that alters water availability and temporal variations in moisture in the root zone will affect productivity.

Owensby et al. (1973) found that routine burning of the prairie was one of the best methods to remove cedars from the landscape, and works especially well on cedars with a height less than 2 m. Grazing and burning, when managed properly, will not only improve range conditions, but can also increase primary production. Owensby (2005) stated that as these management practices become relaxed, many range areas have changed in botanical composition. In areas of Northeast Kansas, the dominant vegetation is changing rapidly from grassland into woodland Briggs et al. (2002). They found a 5.8% yearly expansion rate for eastern red cedar (*Juniperus virginiana*) stands, and state that a tallgrass prairie ecosystem can be converted to a closed-canopy forest in 40 years if burning is discontinued. As eastern red cedar encroachment expands, rainfall reaching the soil surface may decrease due to canopy interception. Water caught by the canopy readily evaporates, reducing runoff and deep percolation as well as water available for soil evaporation and plant transpiration. Thus, the hydrology of the region could be altered by woodland expansion. Thurow and Hester (1997) showed that as juniper cover increased from 0 to 36 % cover, water reaching the soil decreased by 34 %. They concluded that any pasture vegetation conversion from brush to grass will increase soil available water. Gash and Stewart (1977) reported 35 % year-end average interception of bulk rainfall in a mixed stand of Scots (*Pinus sylvestris* L.) and Corsican (*Pinus nigra* var. *maritima* (Ait.) Melv.) pine. Other canopy interception research, such as Llorens et al. (1997), has shown a 24 % interception of bulk rainfall in a *Pinus sylvestris* stand.

The conversion of landscapes from grassland to woodland will affect the surface energy balance by changing the partitioning of available energy. The increasing cedar populations are expected to decrease the fluxes of latent heat and soil heat, while increasing the sensible heat flux. Higher Bowen ratios (i.e., the ratio of sensible and latent heat) are expected from encroached cedar areas based upon physiological differences controlling stomatal conductance of water vapor. While studying a coniferous forest, Lindroth (1985) found that 59% of the net radiation was used for latent heat flux, and 32% was used for sensible heat flux. Of this percentage only 13% of latent heat flux came from the ground vegetation, while 50% of the sensible heat flux came from the ground. Kelliher et al. (1989) and Baldocchi and Vogel (1996) found that latent heat flux above a *Pinus radiata* D. Don canopy was strongly determined by stomatal conductance, vapor pressure deficit, and leaf area index. Kelliher et al. (1993) found that the relationships of surface water conductance and atmospheric saturation deficit were more similar above a coniferous canopy than above a grassland canopy. Baldocchi and Vogel (1996) and McNaughton and Jarvis (1983) explain this trend, and discuss that grassland transpiration is controlled by net radiation, whereas saturation deficit controls forest canopy transpiration.

Soil water is expected to be lower at the woodland site compared with the grassland sites due to canopy interception of precipitation.

Grazing will affect the size and phenology of the vegetation and therefore impact how energy is partitioned between the soil surface and the plant canopy. Li et al. (2006) found that soil moisture was the most important environmental factor controlling the dynamics of evapotranspiration (ET) in grassland ecosystems, and Brye et al. (2000) found that the prairie ecosystems maintained higher soil water contents as compared with cultivated crop lands. When comparing the water balance across ecosystems, Frank and Inouye (1994) found that grasslands

had the highest interannual variability of ET than any other biome including desert and forested ecosystems. Grazing should decrease LAI which will increase soil radiation and decrease canopy intercepted radiation. This should cause increases in both the soil and sensible heat fluxes, while decreasing the latent heat flux. Mid season cattle removal in the moderately grazed intensive early stocked treatment should allow for rapid regrowth of the grazed vegetation. This should decrease the differences between moderately and ungrazed treatments that may exist earlier in the season.

Bremer et al. (2001) found that grazing reduced daily evapotranspiration (ET) up to 40% near the end and soon after the grazing period (e.g., early July). After cattle were removed in mid growing season, ET converged between grazed and ungrazed sites, even though significant differences in LAI existed between sites. They conclude that the younger leaves found at the grazed site likely had lower stomatal resistance and higher transpiration rates than did the ungrazed site. The younger leaves senesced later which resulted in higher ET compared with the ungrazed site later in the season. Virgona and Southwell (2006) found that long duration grazing (5- 8 weeks) resulted in a 22 mm drier soil depth than pasture that did a short duration (2 weeks) grazing. Frank (2003) found that grazing caused a 7% decrease in ET as compared with nongrazed prairie. Naeth et al. (1991) have also documents similar patterns, and add that any differences that may exist in soil water should decrease as soil available water becomes the most limiting factor for plant growth and development. Murphy et al. (2004) recorded daily ET values of 0.2 to 7.2 mm/day and that variations in ET were mostly controlled by solar radiation, herbage mass, vapor pressure deficit and soil water content. The grazed and ungrazed sites used in Bremer's study are two of the sites that will be used in this study. Bremer et al. (2001) only

analyzed a single year, 1999, and thus did not explore how interannual variations in climate affected the response to grazing.

Providing drinking water for grazing livestock is of great concern in the Flint Hills region. Water is often supplied to cattle by pumping from aquifers, lakes, and other water bodies. In areas such as the Flint Hills, the undulating topography can be used to create stock ponds (i.e., earthen impoundments) to for livestock use. Water is then either left open to, or is pumped from these stock ponds to livestock. These stock ponds are filled and replenished by springs, streams, and rainfall. Drought periods can result in dry ponds, thus causing ranchers to provide livestock water by other means. The demanding task of hauling water to cattle can increase labor costs to the livestock producer. Producers need better methods for designing and managing stock ponds, which may result in considerable savings in time and money. Methods similar to those used by Ham (1999, 2004), and Ham and DeSutter (1999) could be used to help estimate or measure the main forms of water loss: evaporation (E), seepage (S), and cattle use (C) from these stock ponds. Utilization of the methods coupled with local weather data, could produce a model to help ranchers predict stock pond capacity and design impoundments of the proper size.

Objectives

- 1) To determine the amount of water lost from a stock pond due to cattle consumption, evaporation, and seepage with the aim of ranking the importance of each form of water loss
 - a) use a micrometeorological raft and ancillary instruments to collect measurements of air temperature and humidity, water surface temperature, wind speed, pond depth, and global irradiance;
 - b) measure or estimate pond water inputs and losses;
- 2) To determine how the removal of biomass by grazing impacts the surface energy and water balances of the tallgrass prairie
 - a) obtain EC data from the grazed and ungrazed pastures;
 - b) compare peak biomass between pastures;
 - c) estimate monthly and growing season ET differences between the grazed and ungrazed pastures;
- 3) To determine how the conversion of a tallgrass prairie to woodland (i.e., eastern red cedar) affects the energy and water balances of the landscape
 - a) obtain and compare energy fluxes from Eddy Covariance (EC) measurements between a woodland and prairie ecosystems;
 - b) collect water balance measurements from the canopy, understory, and soil surface;
 - c) estimate and compare monthly, seasonal, and cumulative energy fluxes and ET between the woodland and prairie;

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CHAPTER 2 - Water Balance of a Stock-Watering Pond in the Flint Hills of Kansas

Introduction

The performance of grazing livestock is strongly affected by access to quality drinking water. Studies have shown that a large fraction of grazing occurs within 365 m (1200 ft) of water (Gerrish and Davis, 1997) and forcing livestock to travel long distances between grass and water decreases performance. Also, any factor that decreases water quality affects weight gain Wells (1995). Thus, watering of livestock has long been a crucial and sometimes limiting aspect of ranching. In areas with adequate precipitation and some relief in topography, water for grazing livestock is often provided by small constructed ponds that are fed by runoff, seeps, or springs. The Flint Hills of eastern Kansas is one such area where earthen ponds are the primary source of livestock water. The Kansas Agricultural Statistics Service reported approximately 1.5 million head of cattle grazed the 3.5 million acres in the Kansas Flint Hills in 2005.

The annual consumption from an individual pond is dependent on the stocking rate, size of the pasture(s) being served by the pond, and stocking duration. Ponds must be designed to collect and store enough water to meet all the consumptive demands of the livestock as well as all other forms of water loss. Essentially, water inputs to the pond must be equal to or exceed all losses; otherwise the depth of the pond will decrease over time and eventually dry up. In many areas of the country, including the Flint Hills of Kansas, the ponds are primarily filled and replenished by runoff from precipitation. According to the National Resource Conservation Service (NRCS, 2005), a pond that is replenished by runoff should have a minimum depth of 1.5

m (5 ft) at the deepest point and a minimal area of 46.5 m² (500 ft²). However, these are general guidelines and mainly address the geotechnical aspects of pond construction. The more difficult question is how should the pond be designed and managed to minimize the probability that the pond will go dry when serving as the primary water source for grazing livestock. Hauling or pumping water to pastures with dry ponds increases costs, increases demands on labor and equipment, and can decrease livestock performance if cattle are forced to travel greater distances to water; all of which decrease profits. Thus, there is merit to closely studying the water balance of livestock ponds to optimize new pond construction and prevent the need to haul water under all but the most severe droughts. Furthermore, the supply capacity of existing ponds could be more closely approximated so that fenced areas and stocking rates could be properly sized for a given pasture-pond combination.

The water balance of a stock-watering pond can be expressed as

$$I + P = E + T + C + S + O + \Delta D \quad [1]$$

where I is inflow into the pond, P is the precipitation falling directly on the pond, E is evaporation, T is transpiration from surrounding vegetation, S is seepage, C is cattle consumption, O is overflow out of the pond, and ΔD is the rate change in depth, with all terms expressed as mm day⁻¹. This equation represents the conservation of mass for a control volume, inputs = outputs + the change in storage, where I and P are inputs and the right-hand side (rhs) represents losses and storage term (i.e., depth change). Unfortunately, the factors affecting the water balance are complex; Table (1) lists over 20 site-specific variables that could alter pond hydrology, factors that includes weather, watershed properties, pond characteristics, and grazing

regime. Predicting runoff into a pond is especially complex, but has been the subject of considerable modeling work (e.g., TR-55 small watershed hydrology model; USDA-NRCS, 2002). In this paper, we will focus on the main forms of water loss; namely E, C and S, as represented on the rhs of Eq. 1.

In the Great Plains, annual evaporation from open water can range from 1.2 to 1.8 m Sophocleous (1998), and probably represents the largest form of water loss for most ponds. Unfortunately, evaporation from ponds has received minimal attention because most studies have focused on larger lakes and reservoirs. Evaporation from stock-watering ponds is complex because the air flowing over the pond never reaches equilibrium with the water surface. Furthermore, the surrounding vegetation and landforms can strongly affect wind flow, a factor that controls the aerodynamic conductance of water vapor between the surface and atmosphere. Measuring and modeling evaporation from small water bodies was evaluated by Ham (1999) when studying the water balance of animal waste lagoons. The modeling approaches based on formulas proposed by Penman (1948) and Priestley and Taylor (1972) are the most common ones applied to ponds (e.g., Steward and Rouse, 1976; DeBruin 1978; Ham, 1999).

Transpiration from trees and vegetation surrounding a pond could be a significant source of water loss at certain locations. For example, during hot summertime conditions, a large cottonwood tree can transpire up to 500 L/day or 132 gallons Schaffer et al. (2000), a value equal to the water consumption of 15 head of cattle. Unfortunately, modeling the water loss from trees and brush around a pond also is challenging. However, methods developed for modeling transpiration from trees in riparian areas could be applied to ponds (e.g., Goodrich et al., (2000)). Ponds with small areas or those with shapes that have a small area to perimeter ratio (i.e., small isoperimetric quotient) will be most affected by shoreline vegetation.

Seepage losses from ponds also have received little study. Ham (2005) measured seepage from 20 animal waste lagoons in Kansas and found an average seepage rate of 1.1 mm/day (0.4 m/yr). While animal waste lagoons are different from ponds, they are a reasonable choice for comparison based on size and depth. Because most ponds do not have a compacted clay liner, we might expect seepage from ponds to be larger than that from lagoons. However, Ham (2005) found that lagoons with no constructed liner still had seepage rates less than 3 mm/day. Also, stock ponds are typically shallower than lagoons so there is less pressure head to drive seepage. In summary, we might expect seepage from many stock-watering ponds to range from 0.9 to 3 mm/day or 0.3 to 1.0 m annually.

The importance of stock-watering ponds to grazing cattle merits additional study of factors affecting pond hydrology on rangelands. In this study, methods similar to those used by Ham (1999, 2005), and Ham and DeSutter (1999) were used to estimate or measure the water balance of a stock-watering pond in the Flint Hills of eastern Kansas. Several meteorological models of pond evaporation also were tested. Results show which components of the water balance are most important and provide background information for an improved design framework for ponds on rangeland. Ultimately, a hydrology based, site specific model of stock-watering ponds could help determine the supply capacity of existing impoundments and improve the design and management of new ponds.

Methods

Site Description

The pond was located in the Rannells Flint Hills Prairie Preserve approximately 9 km south of Manhattan, KS (39°08' N, 96° 32' W, ~340 m above mean sea level). Historical aerial photographs show the pond was built prior to 1971 Jantz et al. (1975). The vegetation in the surrounding pasture was dominated by C₄ grasses, including big bluestem (*Andropogon gerardii* Vitman) and indiangrass *Sorghastrum nutans* (L.) Nash), and has been annually burned each spring for the last several decades. Historically, the pasture had been grazed by stocker cattle in the spring and summer months (May to early October). The soil is classified as a silty clay loam (Benfield series: fine, mixed, mesic, Udic Agriustolls) with slopes of 5-20 percent and has a loamy upland range site classification. The 30 yr average annual precipitation is 880 mm, with 540 mm received between May and September, and Sophocleous et al. (1998) report 1470 mm (58 in) of potential evaporation for the region.

The pond and watershed were mapped with global positioning systems (AgGPS 132 and AgGPS 710, Trimble Navigation Limited, Sunnyvale, CA); areas and slopes were computed using Arcview (9.1, ESRI, Redlands, CA.). When full, the pond had an area of 0.35 ha and was 2.2 m at the deepest point. The pond captured drainage from approximately 25 ha, with the highest point at 425 m and elevation of the pond was 390 m. The average slope (i.e., y-slope) of the drainage was 4.7 degrees. The 65 ha pasture that encompassed the pond was stocked with

yearling steers between May and October. In 2005, grazing livestock were comprised of 12 head of Black Angus (*Bos taurus*) and 36 head of Brahma (*Bos indicus*) cattle with an average initial weight of ~250 kg steer⁻¹ and ~180 kg steer⁻¹ respectively. In 2006, 37 head of Black Angus grazed the pasture with an average initial weight to ~250 kg steer⁻¹.

The cattle were fenced off from the pond and drinking water was supplied to the cattle by a 4.5 m³ (1200 gallon) circular watering trough that was positioned outside the fence below the dam. Routing pond water to a trough located some distance away from the shoreline is becoming common practice to improve water quality Ohlenbusch et al. (1995). The watering trough was supplied from a 1.9 m³ (500 gallon) storage tank positioned on the dam. The storage tank was kept full by a solar-powered pump in the pond that was activated by a float switch. Two flow meters, one analog and one digital, (FTB-6205, and FTB-4707, Omega Engineering, Inc., Stamford, CT) were installed between the solar pump and supply tank to record the volume of water diverted to the stock tank for cattle use.

Water Balance Measurements and Calculations

Evaporation from the pond was measured using the methods of Ham (1999). A meteorological raft (1.5m x 2.0m) was positioned at the center of the pond and carried an infrared thermometer (4000.4ZL Everest Interscience Inc., Tucson, AZ.) for measuring surface temperature, a three-cup anemometer (0301-L, Campbell Scientific Inc., Logan, UT.), and a air temperature and humidity probe, (HMP35-A Campbell Sci. Inc.), all positioned 1 m above the water. Additional instrumentation on the bank of the pond included: a tipping bucket rain gauge (TE-525W, Campbell Sci.), a pyranometer (LI200, Li-Cor Inc. Lincoln NE), and a micrologger (CR10X, Campbell Sci.) for data acquisition.

Hourly evaporation was estimated using the methods of Ham (1999).

$$E' = C_e U_r \rho (q_s^* - q_r) \quad [2]$$

where E' is evaporation rate ($\text{kg m}^{-2} \text{ s}^{-1}$), C_e is the bulk aerodynamic transfer coefficient for vapor (2.8×10^{-3} dimensionless), U_r is the average wind speed at 1 m (m s^{-1}), ρ is air density (kg m^{-3}), q_s^* is the saturated specific humidity at the water surface and q_r is the specific humidity of air at 1 m (kg kg^{-1}). Summing E' over 24 h yields daily evaporation required in Eq. 1.

Depth change in the pond was measured using a float based recorder described by Ham and DeSutter (1999). A linear displacement transducer (LX-PA 50, Unimeasure Inc., Corvallis, OR) with a retractable leader was used to sense changes in water level based on float travel inside a stilling well. The recorder, when logged with the CR10X, had a resolution of 0.24 mm and a full scale range of 1.27 m.

Pond seepage was determined as the difference between depth change and evaporation equation 3, providing I, P, C, and O can be eliminated from the water balance equation.

$$S = \Delta D - E \quad [3]$$

Ham (1999, 2002) demonstrated that the resolution of seepage calculations can be improved when integrating over long time periods ($7 - 10 \text{ d}^{-1}$) during cold weather when E is small. Therefore, seepage was estimated during lengthy dry periods in the winter of 2006 when no cattle were present and no water was entering the pond or passing through the spillway. Seepage calculations for the rest of the year were scaled by pond depth following the approach of Ham (2002, 2005) assuming the hydraulic conductivity of the soil liner did not change over time.

Cattle consumption, expressed in terms of pond depth, was estimated as:

$$C = \frac{V - E_{pan}}{A} \quad [4]$$

where C is cattle water consumption (mm d^{-1}), V is the volume pumped from the pond and delivered to the watering tank ($\text{m}^3 \text{d}^{-1}$), E_{pan} is the evaporation from the watering tank ($\text{m}^3 \text{d}^{-1}$), and A is pond area (m^2).

Evaporation from the surface of the water trough was assumed to be equal to that from a Class-A evaporation pan. Ham (2005) showed that ratio between lagoon and pan evaporation was variable but typically between 0.7 and 0.8 for summer months ($E/E_p \approx 0.75$). Thus E_{pan} was computed from estimates of pond evaporation from Eq. 2, assuming a pan coefficient of 0.75, and adjusting for the area of the watering trough. Because E_{pan} is much smaller than V , errors associated with estimating E_{pan} have little effect on C and the overall water balance

Transpiration from vegetation on the edge of the pond was difficult to approximate. The grasses were thought to have little effect because a zone of bare soil bordered the periphery of the pond. However, there were two mature cottonwood trees (*Populus*) growing on the dam which most likely obtained most of their water directly from the pond. During a low water period, large roots were observed running over the bottom of the lagoon. Using sap flow gauges, Schaffer et al. (2000), showed that transpiration from large cottonwood trees growing in riparian areas in an arid climate was typically 0.2 to 0.5 $\text{m}^3 \text{d}^{-1}$ (52 to 132 gallons per day). Assuming that the maximum water use from the tree at the pond was 0.5 $\text{m}^3 \text{d}^{-1}$ and coincided with the maximum evaporation from the pond, T from a single tree in terms of pond depth can be roughly approximated as:

$$T = \frac{0.5 \cdot E}{A \cdot E_{max}} \quad [5]$$

where E_{max} was the maximum daily pond evaporation observed during the growing season; 10 and 15 mm d⁻¹, for 2005 and 2006 respectively. Transpiration was only calculated between May and September when the tree was fully foliated. While this approach is simplistic, it is a rationale choice given the available data and is better than neglecting T altogether.

On-site instrumentation and supporting calculations provided estimates of most water balance terms, including: P, E, S, C, T and ΔD. Precipitation in the summer of 2006 was below normal and provided long periods when no overflow occurred (O=0). There were three heavy rain events in 2006 that allowed the calculation of inflow from runoff, I, from the residual from Equation 1. Basically, the volume of water entering the pond was calculated from the sudden increase in pond depth and area after a rain storm. Given the land area draining into the pond was known, the percent of precipitation that entered the pond from runoff could be calculated.

Stocker beef cattle are sensitive to heat stress and therefore increase their daily water consumption rates to alleviate the stress Osborne (2003), and Bicudo and Gates (2002, Eq.6). Maximum daily air temperature and relative humidity data were used from a nearby weather station, located at the headquarters of the Konza Prairie Biological Station, to compute a temperature humidity index (THI) for the grazing steers.

$$THI = 0.8t_{db} + RH(t_{db} - 14.4) + 46.4 \quad [6]$$

where t_{db} is the dry-bulb air temperature (°C), and RH is the relative humidity in decimal form. The THI is often used as a heat stress warning system, and provides the producer with values to monitor cattle heat stress. Heat stress levels occur at index values greater than 65, and become more dangerous as the values reach or exceed 70. Comparisons of THI to water

consumption were made on a daily basis to study the response of the grazing cattle during the grazing period.

Modeling Pond Evaporation

One goal of the project was to determine if evaporation from the pond could be estimated using data from a weather station network. The closest weather station was located at the headquarters of the Konza Prairie Biological Station located 7 km west of the pond. Average daily wind speed, air temperatures, and vapor pressure deficits at the pond were compared with the same data collected from the weather station. Data from the weather station were used to calculate daily evaporation using the Penman equation Penman (1948) and the Priestley Taylor model of Stewart and Rouse (1976). The Penman formula was a form of the FAO 56 equation for reference crop evapotranspiration Allen (2005) that had been modified for open-water. After removing the canopy resistance term and using a roughness length of 0.1 cm for a pond sized water body, the resulting equation took the following form

$$E = \frac{0.408\Delta(R_n) + \gamma \frac{418}{T + 273.15} u_2 (e_s - e_a)}{\Delta + \gamma} \quad [7]$$

where E is the reference evapotranspiration (mm d⁻¹), R_n is the net radiation (MJ m⁻² d⁻¹), T is air temperature at 2m (C), e_s is the saturation vapor pressure at air temperature (kPa), e_a is the vapor pressure of air (kPa), u₂ is the wind speed at 2 m (m s⁻¹), Δ is the slope of the vapor pressure curve at air temperature (kPa C⁻¹), and γ is the psychrometric constant (kPa C⁻¹). Details on calculating E using the modified Penman and the Priestley Taylor formulas are provided in Jensen et al. (1989) and Allen et al. (1998).

Results

Weather and Evaporative Demand

In the temperate climate of the central U.S., a large fraction of the annual precipitation and runoff occurs in spring and early summer. In Manhattan, KS, long-term records show the months of May, June, and July account for 56 % of annual precipitation. Thus, stock watering ponds tend to fill to capacity during the spring and early summer and then are depleted by cattle consumption, evaporation, and seepage during late summer and early fall. In 2005, near record precipitation of almost 300 mm fell in June so the pond was filled to capacity at the start of the study (Fig. 1). For the remainder of the 2005 grazing season, precipitation was near normal and reference ET was within 2 % of the historical average (Fig. 1). The pond at the study site and those in the region had good supply of water during the summer and there was no threat of water shortage in 2005. In 2006, precipitation in May and June was 179 mm (83 vs. 262 mm) below normal and spring runoff was insufficient to fill the pond going into the grazing season. Comparisons of reference pond ET in 2006 to the historical average showed that evaporative and reference ET exceeded rainfall by 50 cm. Because of high evaporative demand and below-normal precipitation, water levels in stock watering ponds in the region tended to decline significantly during the summer of 2006 and water shortages were a major concern. However in August, 236 mm of precipitation fell within 20 days (Fig. 1), which refilled the pond.

Pond Depth Changes and Runoff

Pond depth fluctuated over 1.5 m during the 490-day record (Fig. 2). Large runoff events filled the pond to its maximum capacity of 2.2 m in the spring of 2005 and mid-summer of the 2006, while the low water mark of 0.6 m occurred in August, 2006 at the end of a summer drought. Because rainfall events in the High Plains are infrequent and episodic, there were periods between inflow events that showed a steady decline in depth. The largest of these drawdown periods, a 0.8 m decline, occurred between June 14th and August 13th, 2006 when the pond got so low the water supply for the cattle was almost disrupted. Filtering out the few instances of precipitation, runoff, and overflow during the study allowed calculation of daily water losses (S+E+T+C) solely from the change in depth measurements (Fig. 3). Data show annual cycle of water loss with peak values near 17 mm d^{-1} during both years. Average summer (June 21 to September 22) loss rates were 14.2 and 14.6 mm d^{-1} for 2005 and 2006, respectively. There was a rapid decline in the rate of water loss starting in October 2005 with lowest values of 1.5 mm d^{-1} occurred during an unusually cold December in 2005. Rates of water loss increased steadily during the winter and spring of 2006 and varied from 8 to 18 mm d^{-1} throughout the summer depending on weather conditions. In 2006, the rate of loss during September was greater than during July, even though evaporative demand was greater in July. This suggested that the increase in depth and area following the August rains may have increased losses from S and possibly T. Despite the weather-induced variability in Fig. 3, these data show the utility of simple depth measurement when addressing pond hydrology. If depth time series were collected for several years in multiple ponds in a region, it would be possible to derive a good “rule of thumb” estimates of daily and monthly loss rates; numbers that might aid pond design and management.

Three heavy rains received during mid to late August 2006 caused significant runoff and provided a good opportunity to see how well pond inflow could be predicted with a simple model. Actual runoff from these events was measured from pond depth changes and surface area (Q_{meas}), while runoff also was modeled using the NRCS curve number method (Q_{mod}) (Table 2). Prior to the first runoff event, the region had received minimal precipitation resulting in drier than normal soil moisture profile. On August 14, 81.9 mm of rain was received over 7.5 hours. Modeled runoff from this event was 26.5 mm, while only 8 mm of runoff was measured by pond measurements. Results from event 2 were more comparable at 8.0 mm and 6.8 mm for Q_{meas} and Q_{mod} respectively. Event 3 resulted in slightly lower similarity with 10.7 mm measured and 6.8 mm being modeled. Calculations showed that dry soils retained nearly 60% ($I/P = 0.6$) of the precipitation that fell on the initial heavy rain on August 14, and explain the dissimilarity between Q_{meas} and Q_{mod} during the first event. Initial abstraction (I_a), the runoff curve number model parameter that accounts for infiltration and capture before runoff begins, is highly variable and depends upon antecedent soil moisture and soil cover (USDA-NRCS, 2002). Measured and modeled runoff from events 2 and 3 may have been more similar than event 1, because of the increase in soil water content from the first precipitation event. Results in Table 2 suggest it will be challenging to estimate runoff and pond inflow in the Flint Hills of Kansas with any accuracy unless antecedent soil water content is included in the modeling framework.

Seepage

The seepage rate from a pond is dependent on the liner permeability, liner thickness and hydraulic head. All of these parameters can vary spatially and with pond depth as the submerged area changes. The apparent whole-pond seepage rate was calculated during a 20-day study in December and January of 2005-2006. Winter is the best time to conduct the test because Ham

(2000) showed that the uncertainty seepage estimate is lowest during periods of low evaporation. Also there was no inflow or outflow from the pond during this period so that the change in depth was solely from evaporation and seepage Ham (1999). Figure 4 shows cumulative depth change and total evaporation over the 20 day study the difference in the two totals representing seepage. The calculated seepage rate was 2.6 mm d^{-1} ; a value is consistent with the minimum wintertime rate change in depth data observed in Figure 3. Assuming an apparent liner thickness of 30 cm and a average pond depth of 1.5 m, the whole-pond hydraulic conductivity was computed as $7.59 \times 10^{-7} \text{ cm s}^{-1}$ following the procedures of Ham (2005). While the thickness of the liner was not known, it was important to parameterize the seepage in terms of permeability so that seepage could be scaled during the rest of the study as the depth of the pond changed. Figure 5 shows the calculated seepage rate for the entire study period as calculated from hydraulic conductivity and pond depth. On average, seepage was 2.6 mm d^{-1} , but ranged from 3.7 to 1.4 mm d^{-1} . These seepage rates and the ponds hydraulic conductivity are about three times higher than those from earthen basins with compacted soil or clay liners Ham (2002).

Evaporation

Evaporation was highly variable but demonstrated clear seasonal trends (Fig. 6). The average evaporation rate for the summer (June 21 to September 22) was 5 and 7 mm d^{-1} for 2005 and 2006, respectively. Lower VPDs and wind speeds in 2005 resulted in less evaporative demand compared to drier and windier conditions in 2006. Peak evaporation rates of 17 mm d^{-1} occurred in July, 2006 when wind speeds were over 5 m s^{-1} and air temperatures exceeded 38 C. Advection of sensible heat from the surface boundary layer provided extra energy for evaporation. Examination of the 2006 grazing period (May to October) showed a mid-season trend developed in which evaporative demand decreased after August 8. Between June 15th and

August 8th, 490 mm of evaporation was recorded as compared to 247 mm for the later half of the study. This pattern was caused by a change in weather patterns that decreased evaporative demand (Fig. 1). Examination of a full year of data (October 2005 to October 2006) gave an annual evaporation of 1472 mm, which was comparable to Penman open-water evaporation of 1448 mm calculated from long-term weather records for this region of Kansas Sophocleous (1998). Rainfall during the same period was 708 mm.

Cattle Consumption and Transpiration

Consumption of water by cattle and transpiration were components of the pond water balance between May and October, a period that covered both the grazing season and the time of active plant growth (i.e., green leaves). Results show that both C and T were less than 1.0 mm d⁻¹ and were very small components of the water balance (Fig. 7). Both decreased significantly after August 14, 2006 when heavy rains caused ponding of water at other locations in the pasture and the cattle drank from multiple sources. Evaporative demand also decreased during this period

The cattle consumed between 12 and 46 L/day/animal during the first part of the 2006 grazing season when the pond was the only source of drinking water. On average, consumption was 30 L day⁻¹ animal⁻¹, which was comparable to values in the literature for cattle of this weight (Bicudo and Gates (2002), Gerrish and Davis (1997), and Osborne 2003). Water consumption was only loosely correlated with temperature humidity index (THI). Regression of consumption vs. THI (Eq. 6) resulted in the equation $C = 45.6 + 1.1 \cdot \text{THI}$, $r^2 = 0.39$. Therefore, factors other than THI were governing water consumption. One controlling factor may have been the forage water content (Bartholomew et al. (2001)). Lack of precipitation likely decreased forage water content, thus increased drinking water consumption needed to meet the cattle's water demand.

Partitioning Water Losses during the Grazing Season

A summary of the monthly water losses by component over the five month grazing season in 2005 and 2006 showed that evaporation was main source of water loss, accounting for 57 to 77 % of the total (Table 3). On average, evaporation was 64 percent of the total water loss, seepage was 31 percent, while cattle consumption and transpiration accounted for the remaining 5 percent. July was the month with the greatest water loss in both years; 277 and 358 mm in 2005 and 2006, respectively. Even during these months, cattle consumption still only 4 % of the total loss. Because evaporation was such a large fraction of the water balance, management practices that might reduce losses from the pond are limited. In this case, one option would be to dewater the basin and excavate to create greater depth and install a compacted soil liner. A deeper pond would provide more storage without increasing the surface area for evaporation. While increasing depth will raise hydraulic head and increase seepage, installation of a compacted clay liner would moderate this effect and likely still reduce seepage rates to about one half or one third of those observed in Table 3 Ham (2002).

Evaporation Modeling

Another goal of the study was to determine if data from local weather stations could be used to estimate the monthly evaporation from the stock watering ponds. One question was how comparable were weather conditions at the pond to those at a weather station located 7 kilometers away. Weather conditions at the pond were compared with data from the nearby Konza Prairie Biological station from June through August, the three months with the greatest evaporative loss. On average, air at the pond was about 2.0 C cooler and VPD was 0.2 kPa lower at the pond (Table 4). Cooler and more humid conditions at the pond are not surprising considering the latent heat flux from the water and the higher soil moisture contents in the

lowland landscape surrounding the pond. Surprisingly, wind speeds at the pond was only about 10% lower than the weather station data even though the Konza weather station was in an upland location and the pond was at the bottom of the catchment (Table 4). A funneling effect from drainage landforms may have amplified wind speeds at the pond and compensated for its lowland location. Assuming radiation was identical between the two sites, sample calculations of reference evaporation (Eq. 7) using weather data from the pond and Konza stations showed that average evaporative demand was 6 mm d^{-1} and 7 mm d^{-1} at the pond and Konza station; respectively, for the months of June through August.

Comparisons of the weather data indicated adequate agreement between data collected from the pond and weather station (Table 4), which permitted the use meteorological models to estimate evaporation from the pond. Relatively good agreement was observed between evaporation measured by the pond instrumentation and that calculated from the Penman and Priestley Taylor models using data from Konza as input. Compared to measured values, both models overestimated evaporation in the summer of 2005 and underestimated evaporation during the hottest months of 2006 (Fig. 8). When only the grazing season was considered, May to October, the Penman and Priestley Taylor models underestimated evaporation by 13 and 14 %, respectively in 2006, and overestimated evaporation by similar amounts in 2005. Conditions in 2006 were warmer and windier compared to 2005 (Table 4). The Priestley Taylor model does not use wind speed or VPD as inputs and neither model accounts for horizontal advection, a factor that could significantly increase evaporation from small water bodies Webster and Sherman, (1995). During the fall and winter, the Penman model produced slightly higher estimates for evaporation than the Priestley Taylor formula and was typically in better agreement with the measured data during this period. Using the pooled monthly evaporation data from 2005 and

2006, there was good agreement among the measure and modeled values (Fig. 9). Regression of monthly measured and modeled evaporation from both years yielded an R^2 of 0.81 and 0.86 for the Priestley-Taylor and Penman formulas, respectively. The slopes from both models ($0.97E_{\text{Priestley-Taylor}}$ and $0.95E_{\text{Penman}}$) were slightly less than unity and can be used as pond specific coefficients (i.e., much like crop coefficients for ET) to make predictions of actual pond evaporation from calculated values of reference evaporation (e.g., Eq. 7).

DISCUSSION

The study demonstrated that it is possible to monitor the water balance of the stock water pond for extended periods. Results showed that evaporation accounted for 64 % of the total loss of water during the grazing period. Peak evaporation rates of 10 to 17 mm d⁻¹ were common in the months of July and August. Unfortunately, there is little that can be done from a management perspective to decrease evaporation from stock ponds other than perhaps making ponds deeper with less surface area. Floating synthetic covers, like those used on some waste lagoons, might be cost effective for ponds in remote, arid locations. Seepage was the next most critical form of loss accounting for 31 % of the total. Seepage losses from this pond could likely be reduced by one half to one third of the current rate with the installation of a compacted clay liner. However, the cost benefit ratio of such an investment would need to be considered. Cattle water consumption was only 3% of average loss from the pond and reached a peak of 46 L d⁻¹ animal⁻¹ during July, 2006. In this case, the pond could have supported a much larger number of cattle. Though transpiration ranked last in the amount of water lost, it was just slightly lower than cattle consumption at this pond. Large phreatophytes, like the *Populus* trees, and other surrounding woody vegetation could be removed for that reason to increase the amount of potential water use for livestock.

While not the focus of this study, predicting inflow into a pond in convoluted, hilly terrain like the Flint Hill of Kansas continues to be a challenge. Most of the inflow from runoff occurred during a few infrequent storms. As expected, inflow in the pond was highly dependent on soil moisture conditions at the time of precipitation. The NRCS curve number method for modeling of runoff did not provide accurate estimates of inflow to the pond when soil conditions were dry at the start of precipitation. Modeling approaches that include the impact of antecedent soil water content on runoff will be required in a comprehensive model of pond hydrology Silveira et al. (2000).

A goal of this research was to determine if pond evaporation could be modeled using data from weather station networks. Monthly comparisons of average air temperature, VPD, and wind speed between the stock watering pond and the Konza HQ weather station had sufficient similarity to allow use in meteorological modes. On average, the Priestley-Taylor and Penman equations slightly overestimated evaporation by 3 and 5 %, respectively. These models are used to compute reference pond evaporation and then the result multiplied by a “pond coefficient” to estimate actual pond evaporation. Results showed that the Penman model with a multiplier of 0.95 would have predicted evaporation to within ± 6 % for any month over the grazing season (May to October).

IMPLICATIONS

Ultimately, the goal is to locate and design ponds for a given pasture and grazing regime that can provide season long drinking water on all but perhaps the driest of years. This study showed that evaporation can be modeled and other forms of loss quantified with a relatively simple set of measurements. A clear need is to collect pond water balance data at multiple locations throughout a region to quantify the site-to-site variation. One interesting finding was

that much could be learned solely from the time series of pond depth (e.g., Figure 2). Because inflow events tend to be episodic in the Great Plains, it was possible to quantify the rate of loss (the sum of E, S, C, and T) for most of the year from the slope of the depth vs. time curve. Furthermore, during the winter, the rate change in depth provided an approximation of seepage. Finally, the sudden increases in depth following a rainfall provided a measure of runoff. Thus, depth measurements alone coupled with a few other measures of catchment area, pond dimensions, etc. provides the researcher with detailed knowledge of site specific pond hydrology. Thus, if multiple ponds in a region were equipped with high-resolution depth recorders and recording rain gauges, much could be learned about pond hydrology with minimal expense and effort. This research as well as the combined findings from many other research projects clearly demonstrates that adequate technology and knowledge is available to provide site-specific designs for stock watering ponds and livestock watering strategies in the Great Plains. In areas where livestock drinking water often becomes limiting, having properly designed and managed stock watering ponds could have significant economic benefits.

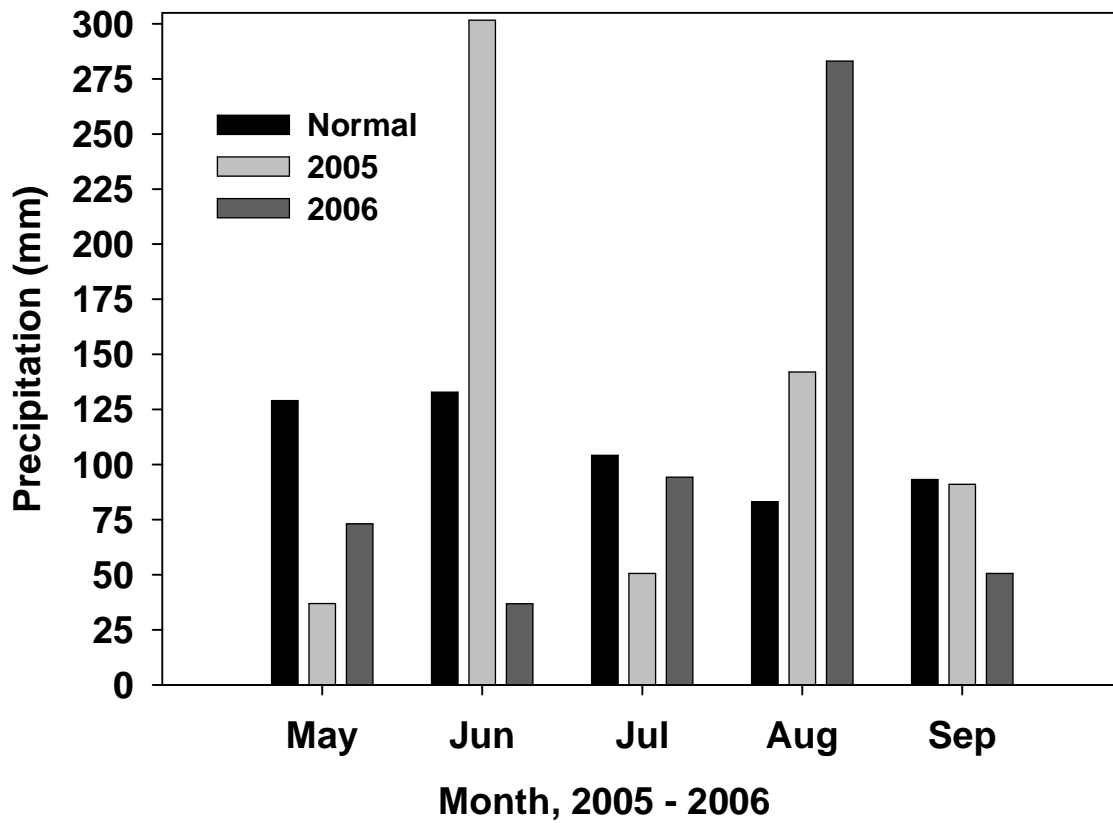


Figure 2-1. Comparison of precipitation during 2005 and 2006 to the 30-yr (1971-2001) average for Manhattan, Kansas.

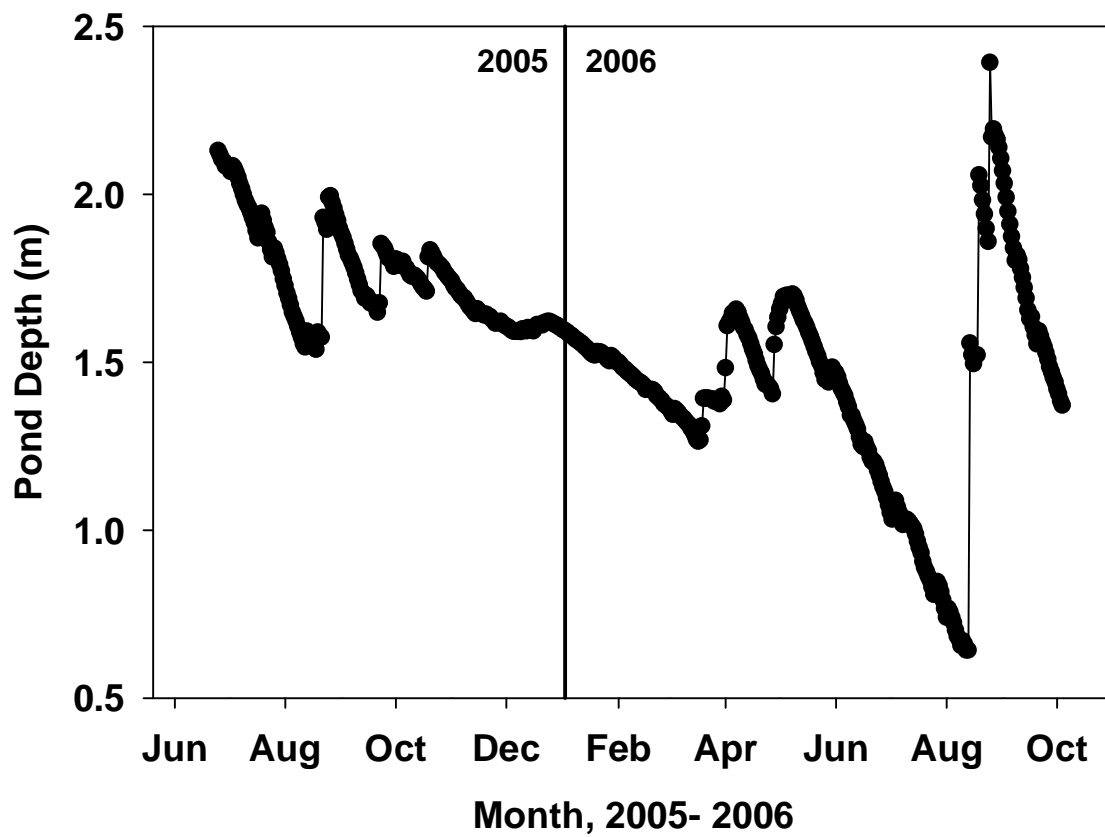


Figure 2-2. Depth of the pond at the deepest point as measured by the depth recorder in 2005 and 2006.

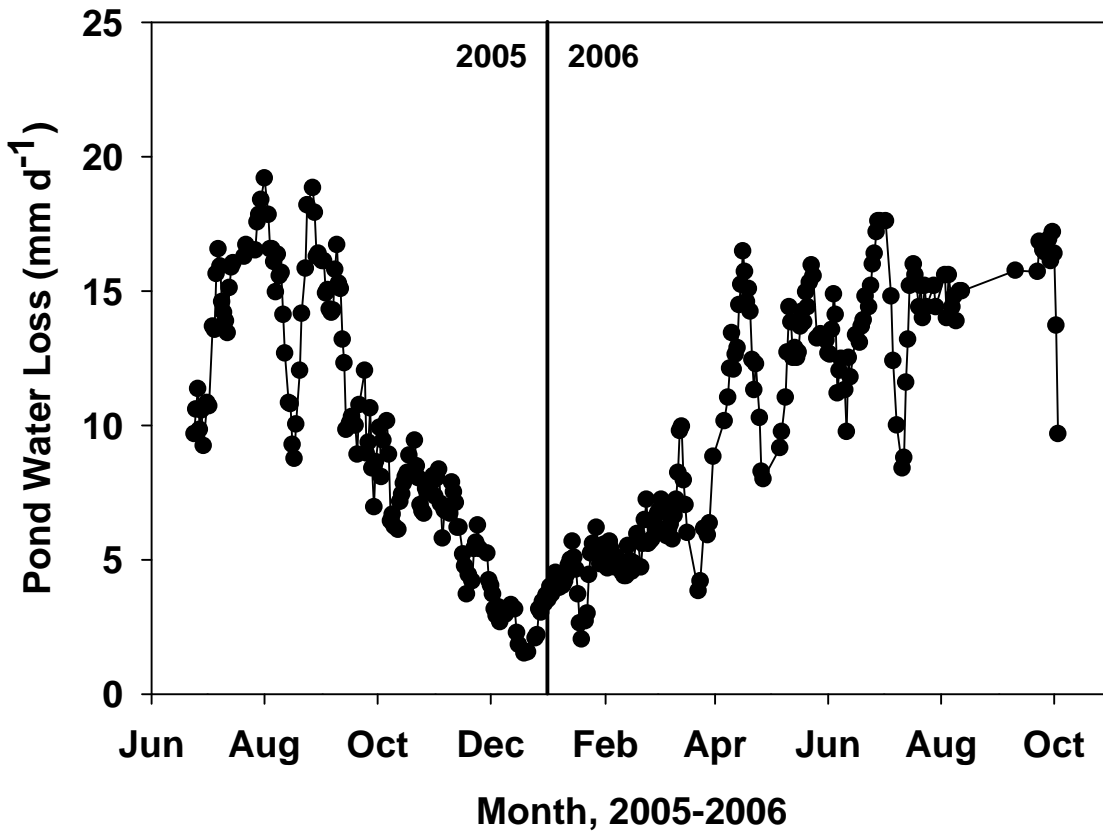


Figure 2-3.Total daily pond water losses for the entire study as measured by the floating depth recorder. The graph represents the change in pond depth, or the combined losses from E, T, C, and S over time.

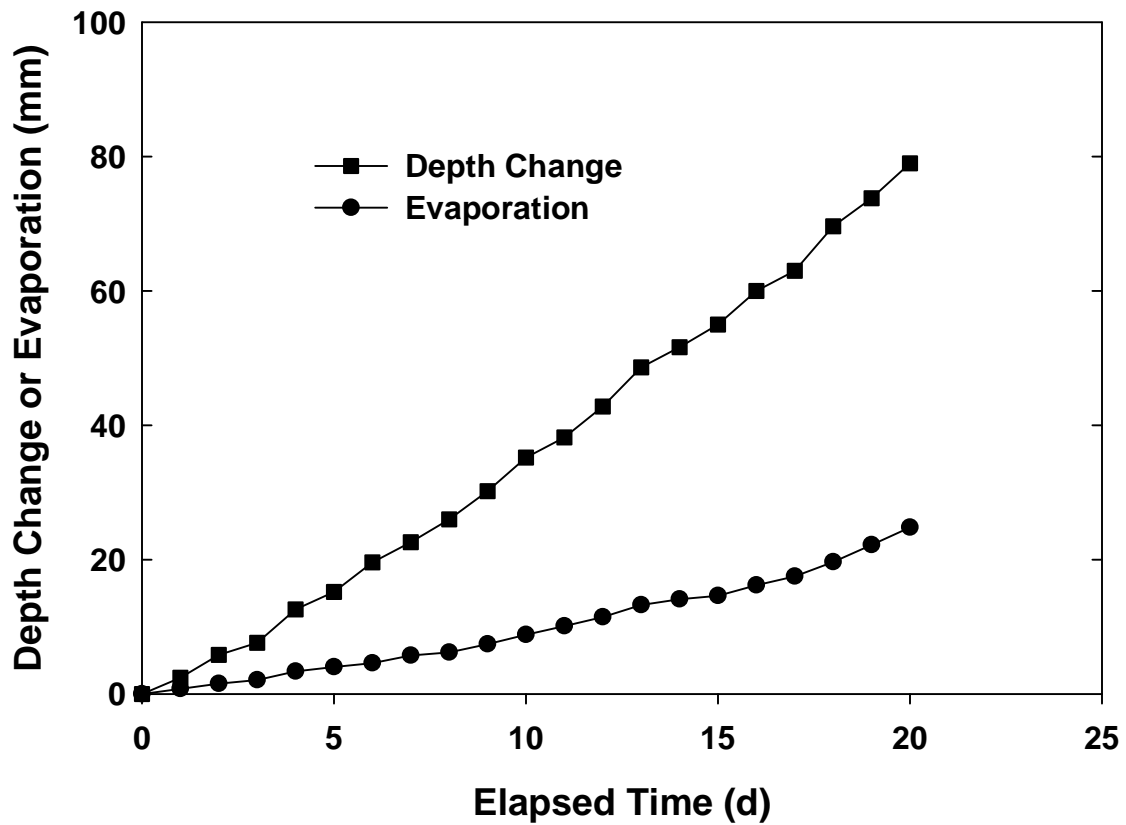


Figure 2-4. Change in depth and evaporation from the pond during a 20-day seepage test between December 27, 2005 and January 15, 2006. Seepage rate was calculated as the difference between total depth change and cumulative evaporation over time. The apparent seepage rate for the test was 2.6 mm d^{-1} .

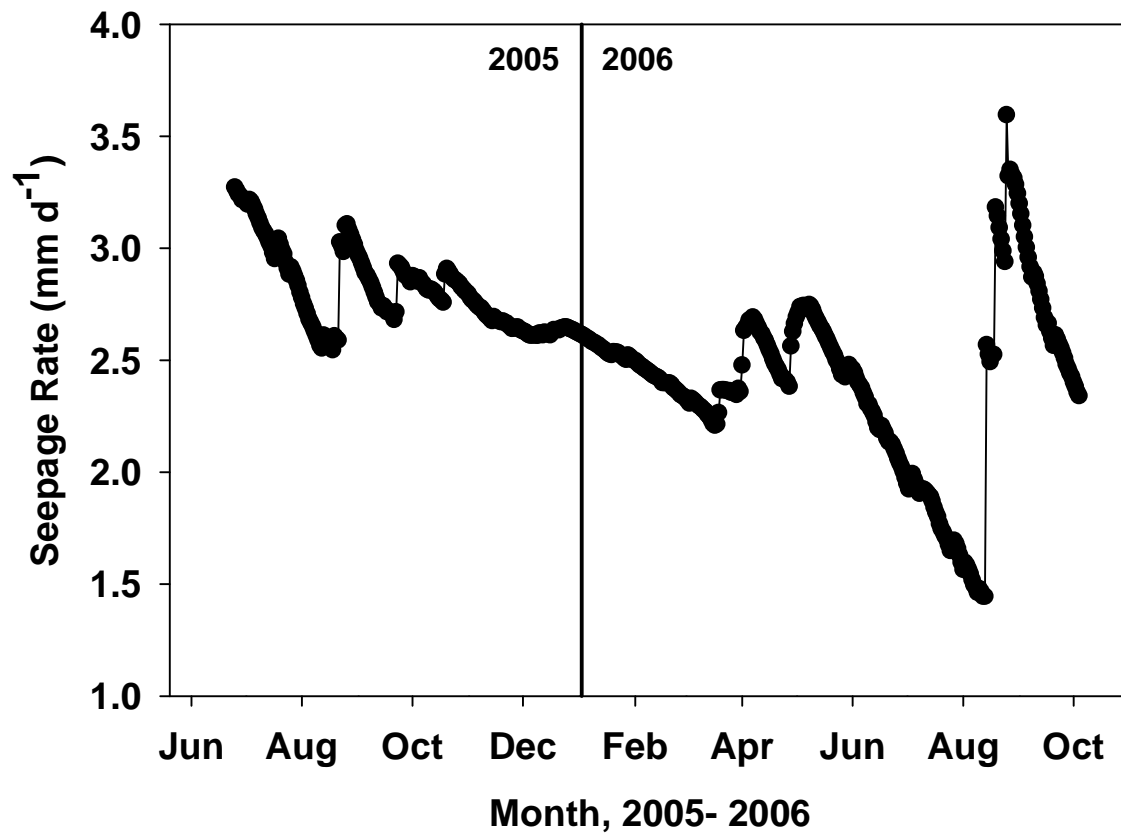


Figure 2-5. Fluctuations in the apparent seepage rate for the stock pond over the entire study period. Since seepage is influenced by hydraulic head pressure at the soil liner, seepage rates fluctuate as pond depth changes.

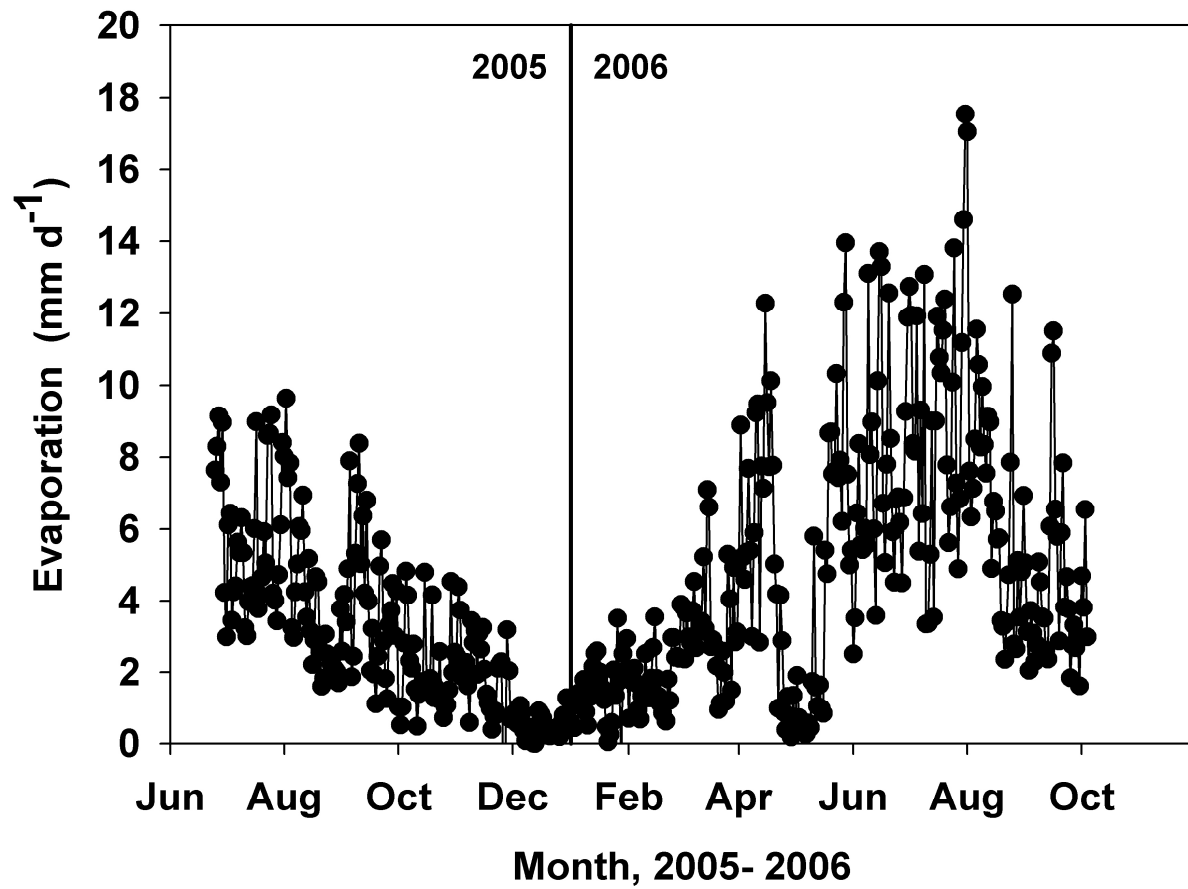


Figure 2-6. Evaporation from the stock-watering pond during the grazing period of the study. Evaporation was measured using instrumentation on the meteorological raft floating on the center of the pond.

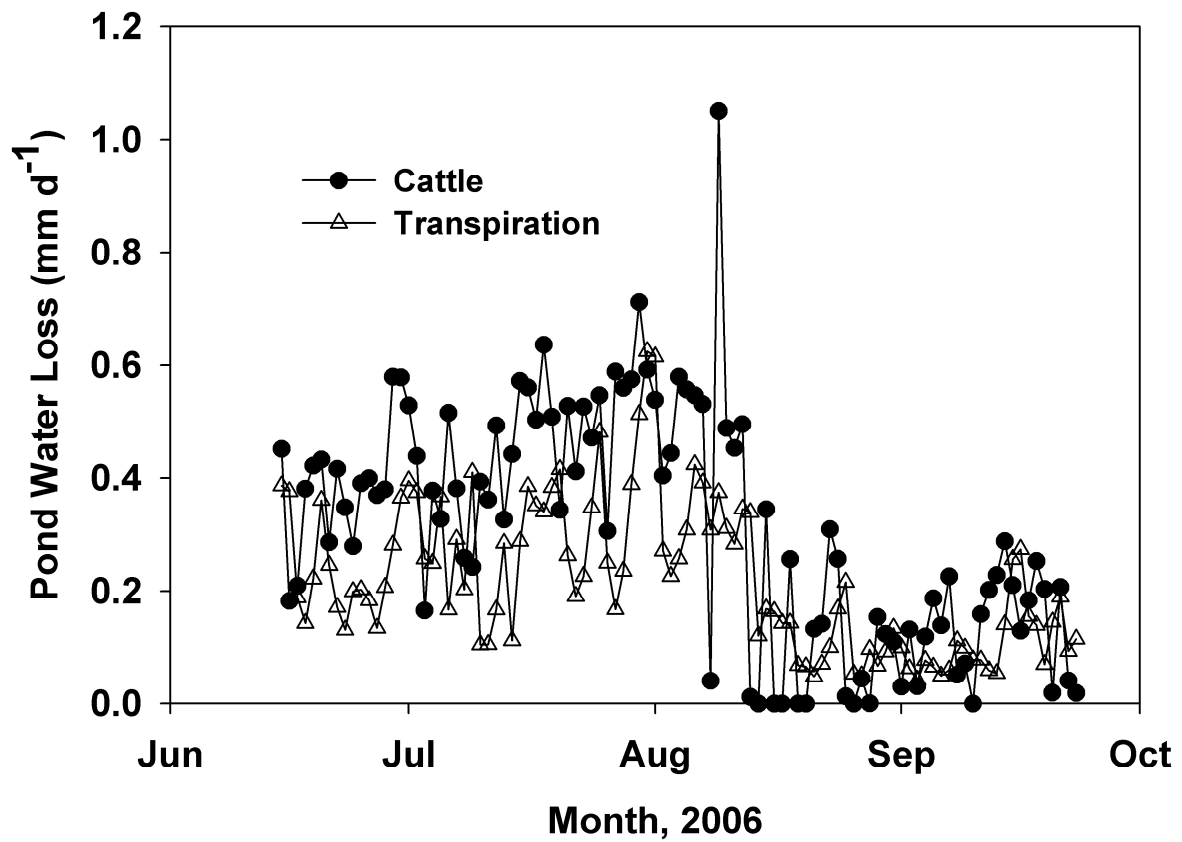


Figure 2-7. Cattle consumption and transpiration from the pond during the 2006 grazing season. Data are expressed in terms of pond depth.

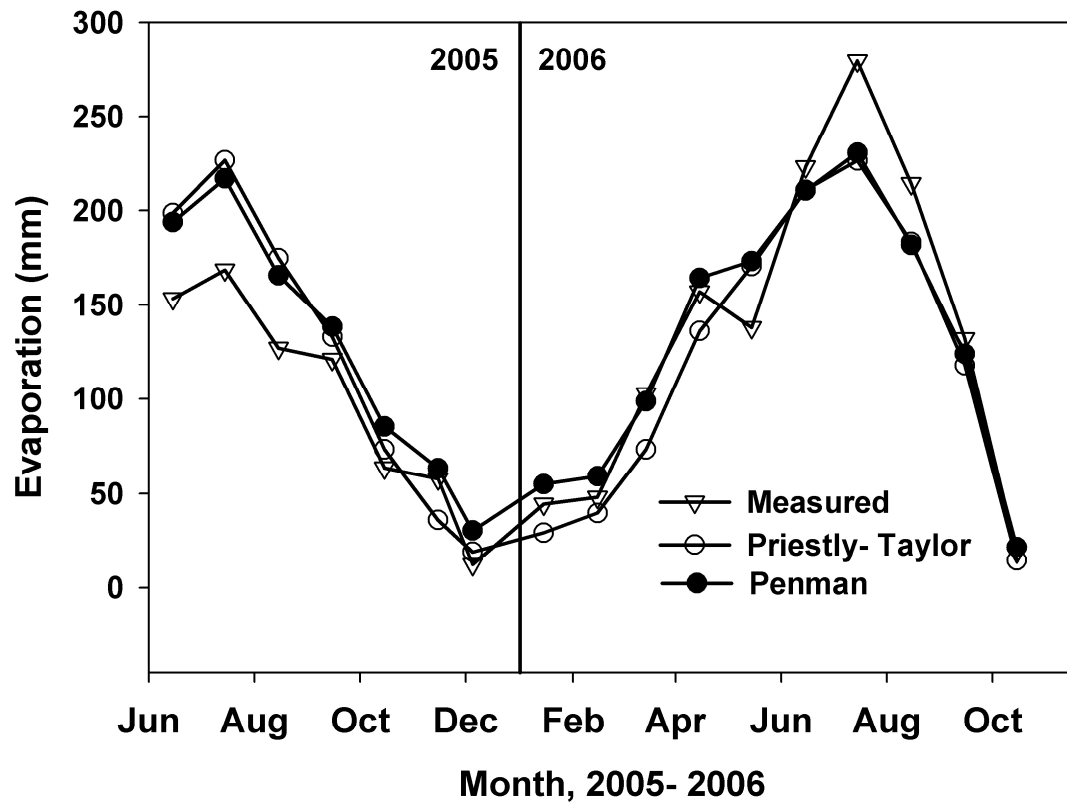


Figure 2-8. Comparison of monthly evaporation measured from pond instrumentation to estimated evaporation derived from the Priestley-Taylor and Penman models using local weather data from the Konza Prairie Biological Station.

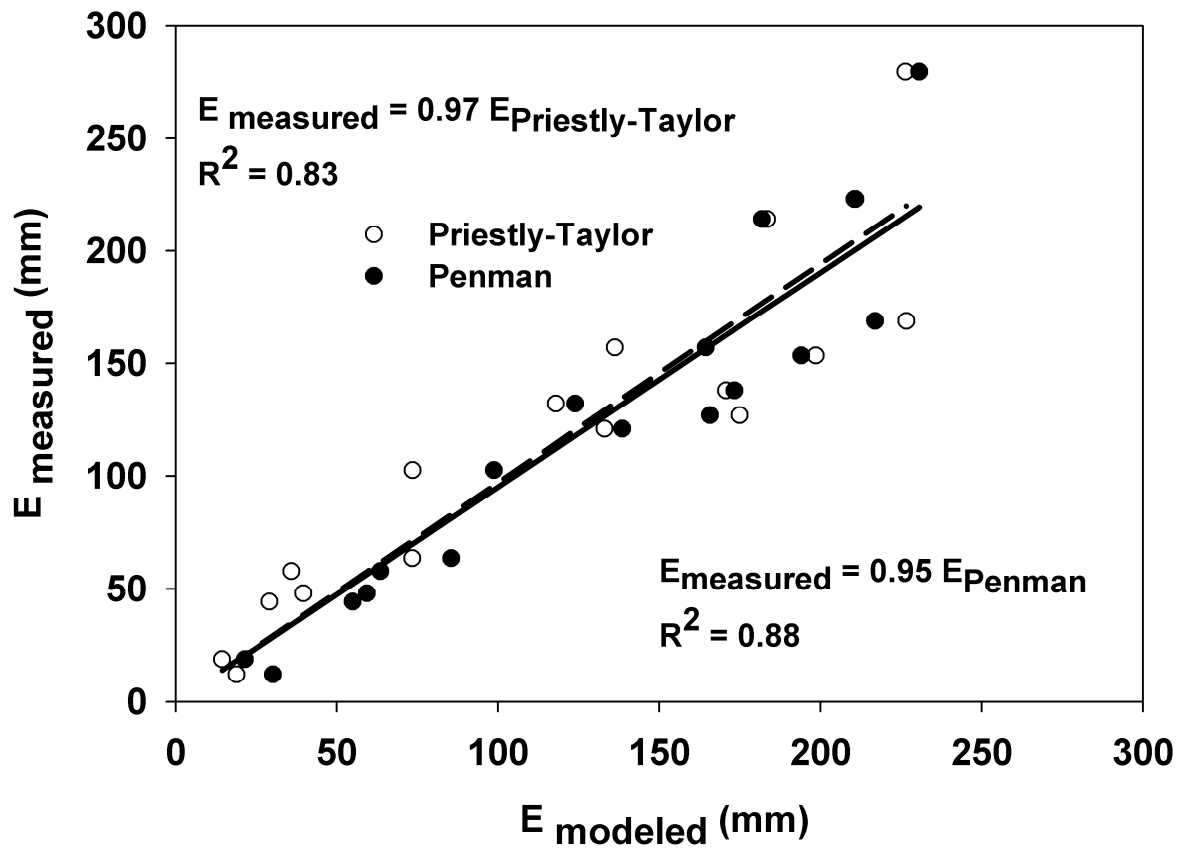


Figure 2-9. The correlation between measured monthly evaporation using the bulk transfer equation to the Priestley – Taylor (---) and to the Penman (—) evaporation models. Modeled values are from weather data obtained from the Konza Biological Station, and measured values are from the actual evaporation measured from raft instrumentation on the pond.

Categories	Water Balance Parameters	Governing Variables
Weather and Watershed	I, P, E, T	Radiation, wind, humidity, temperature, precipitation, pond area, drainage area, vegetation type and size, antecedent soil moisture, soil type, slope and topography
Stocking Management	C	Stocking density, cattle size, breed, forage quality, and weather
Pond Design	S, O	Pond area, water depth, soil liner properties, degree of sedimentation, and water table depth

Table 2-1. Factors affecting the pond water balance (see Eq. 1).

Table 2-2. Comparison of the measured pond inflow to modeled runoff using the NRCS curve number method. *

Event	Period (DOY)	P (mm)	ΔD (mm)	$A_{P\text{ ave}}$ (m²)	$Q_{\text{ meas}}$ (mm)	$Q_{\text{ mod}}$ (mm)	$I_{\text{ fitted}}$ (mm)	I/P (mm)
1	226	81.5	936.6	1561.1	8	26.5	50.4	0.6
2	230-231	46.3	550	2635.3	8	6.8	15.3	0.3
3	237	44.9	716.5	2722.5	10.7	6.8	8.1	0.2

*Data shown are from 3 separate precipitation events that occurred during middle to late August, 2006. Where P, precipitation; ΔD , change in depth; $A_{P\text{ ave}}$, average pond area; $Q_{\text{ meas}}$, measured runoff into the pond; $Q_{\text{ mod}}$, modeled runoff from curve number; $I_{\text{ fitted}}$, initial abstracted precipitation; I/P, percent of P abstracted by the landscape.

Table 2-3. The total monthly losses of water from the stock-watering pond during the 2005 and 2006 grazing period.

Date	Evaporation (mm) (%)	Seepage (mm) (%)	Cattle (mm) (%)	Transpiration (mm) (%)	Total (mm)
June, 2005	153 (58)	96 (36)	12 (5)	3 (1)	264
July, 2005	169 (61)	94 (33)	11 (4)	3 (2)	277
August, 2005	127 (57)	86 (39)	8 (3)	3 (1)	224
September, 2005	121 (56)	85 (41)	3 (2)	3 (1)	212
May, 2006	138 (61)	81 (36)	4 (2)	3 (1)	226
June, 2006	223 (74)	61 (21)	10 (3)	6 (2)	300
July, 2006	279 (77)	56 (16)	14 (4)	9 (2)	358
August, 2006	214 (71)	74 (24)	8 (3)	6 (2)	302
September, 2006	132 (60)	82 (37)	3 (2)	3 (1)	220
Average	173 (64)	79 (31)	8 (3)	4 (2)	265

Table 2-4. Monthly weather parameters recorded from the weather station located at the Konza Prairie Station and the stock-watering pond. Shown in the table are the average air temperature, vapor pressure deficit (VPD), and wind speed for three months of the pond study.

Date	Tempearture (°C)		Vapor Pressure Deficit (kPa)		Wind Speed (m s⁻¹)	
	KZ	Pond	KZ	Pond	KZ	Pond
June, 2005	24.3	24.1	1.1	0.9	3.6	2.8
July, 2005	26.1	25.4	1.4	1.1	2.8	2.1
August,	25.3	24.4	1.1	0.9	2.4	1.9
June, 2006	23.9	21.2	1.3	1.1	2.9	2.9
July, 2006	28.0	24.6	1.8	1.5	2.8	2.9
2006	26.3	22.7	1.3	1.0	2.6	2.6
Average	25.7	23.7	1.3	1.1	2.8	2.5

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CHAPTER 3 - The Effect of Grazing on Evapotranspiration and Net Carbon Exchange in Tallgrass Prairie

Introduction

The Flint Hills region in eastern Kansas is part the largest unbroken tracts of native tallgrass in North America. The tallgrass prairies serve as a building block to the stocker cattle industry. According to the National Agricultural Statistics Service, approximately 1.5 million beef cattle were listed on inventory for the Tallgrass prairie region of Kansas in 2007. In general, the tallgrass prairie ecosystem is water limited Briggs and Knapp (1995); thus any management factor that alters water availability and temporal variations in moisture in the root zone will affect productivity. Li et al. (2006) found that soil moisture was the most important environmental factor controlling the dynamics of evapotranspiration (ET) in grassland ecosystems, and Brye et al. (2000) found that the prairie ecosystems maintained higher soil water contents as compared with cultivated crop lands. When comparing the water balance across ecosystems, Frank and Inouye (1994) found that grasslands had the highest interannual variability of ET than any other biome including desert and forested ecosystems. Bremer et al. (2001) compared ET from adjacent grazed and ungrazed pastures in tallgrass prairie for a single year using micrometeorological techniques. Their grazed site was managed with an intensive early stocking regime Smith and Owensby (1978), Owensby et al. (2006), and Owensby et al. (2008) where cattle are initially stocked at higher densities but removed mid summer to allow for rapid regrowth of the grazed vegetation. This is one of the most common grazing regimes in the Flint Hills. They found that IES grazing reduced season long ET (DOY 128-DOY202) by 6 % ,

but could cause reductions in daily ET by up to 40% near the end and soon after the grazing period (e.g., early July). After cattle were removed in mid growing season, ET converged between grazed and ungrazed sites, even though differences in LAI existed between sites. They conclude that the younger leaves found at the grazed site likely had lower stomatal resistance and higher transpiration rates than did the ungrazed site. The younger leaves senesced later which resulted in higher ET compared with the ungrazed site later in the season. Virgona and Southwell (2006) found that long duration grazing (5 to 8 weeks) resulted in a 22 mm drier soil depth than pasture that had been short duration grazed (2 weeks). Frank (2003) found that grazing caused a 7% decrease in ET as compared with a nongrazed prairie. Naeth et al. (1991) also documented similar patterns, and added that any differences that may exist in soil water should decrease as soil available water becomes the most limiting factor for plant growth and development. Murphy et al. (2004) recorded daily ET values of 0.2 to 7.2 mm/day and that variations in ET were mostly controlled by solar radiation, herbage mass, vapor pressure deficit, and soil water content. Grazing affects the size and phenology of the vegetation and therefore impacts how energy is partitioned between the soil surface and the plant canopy. Grazing decreases LAI which will increase soil radiation and decrease canopy intercepted radiation, causing decreases in latent heat flux from the canopy but increases in soil surface evaporation, especially after precipitation when the soil is wet. Because increased soil evaporation may compensate for LAI-induced reductions in transpiration, one cannot always assume that grazing will cause a reduction in ET. Day and Delting (1994) found that removal of leaf area by grazing caused an increase in grass canopy temperature and evaporative demand, but the negative effects were balanced by an increase in available soil water as compared with ungrazed sites.

Net carbon exchange (NCE) may also be affected by the reduction of biomass and leaf area. Frank (2004) found that cooler than normal temperatures and below normal precipitation can limit the gains of CO₂ in grazed systems. Frank also found that root biomass is often less in grazed systems than ungrazed pastures. Owensby et al. (2006) found that grazing can reduce NCE by removal of biomass, but NCE is often higher in grazed pastures later in the season since regrowth results in younger and more efficient leaf tissues. LeCain et al. (2000) found that carbon exchange was related to green leaf area, and that reductions in green leaf area can reduce carbon exchange. They also found that litter removal by grazing increased light penetration and soil temperature, often resulting in increased growth and carbon fluxes as compared with ungrazed pastures. Suyker and Verma (2001) also observed that increased light penetration and green leaf area resulted in higher carbon exchanges. They also noticed that carbon exchange rates increased as plant water stress decreased. Baldocchi (1994) used the ratio of NCE:ET as a surrogate for water use efficiency. This ratio of NCE to ET is a way of looking at how grazing by removal of biomass and green leaf area may effect carbon water dynamics.

Regardless of the mechanisms involved, quantifying the effect of grazing on ET and NCE is essential for understanding the hydrology and carbon dynamics of rangelands. Evapotranspiration is typically the largest term in the water balance other than precipitation and NCE is an indication of primary productivity. Thus understanding, grazing effects on ET are needed to determine how range management might effect runoff and drainage, pond hydrology Duesterhaus et al. (2008), soil erosion and water quality Dahlgren et al. (2001), and potential range responses to climate change Parsons et al., (2001); Asner et al., (2004). Combining studies of ET with measurements of NCE provide a method for evaluating the effect of grazing on carbon - water relationships (i.e., apparent water use efficiency). Both ET and NCE are strongly

affected by interannual variations in precipitation and evaporative demand. Thus, a multi-year evaluation of how grazing effects ET and NCE is needed. This study compares ET and NCE measurement from adjacent grazed and ungrazed pastures for three years using long-term eddy covariance flux measurements.

Materials and Methods

Study sites included a grazed and an ungrazed tallgrass prairie located in the Flint Hills of northeastern, Kansas USA. The prairie sites were on adjacent pastures that were part of the Rannells Flint Hills Prairie Preserve, approximately 5 km south of Manhattan, KS (lat 38° 08'N, long 96° 32'W, and 380 m above mean sea level). The vegetation was dominated by native C₄ grasses, including big bluestem (*Andropogon gerardii* Vitman), Little bluestem (*A. scoparius* Michx.) and indian grass (*Sorghastrum nutans* (L.) Nash). The prairie sites have been annually burned each spring for the last several decades and the ungrazed pasture has not been grazed since 1997. Peak above-ground biomass for ungrazed sites in the study area typically range from 180 to 760 g m⁻² Briggs and Knapp (1995), while peak leaf area indices range from 1.24 to 3.99 m² m⁻². The grazing treatment used was the moderately grazed intensive early stocked (IES) pasture management plan proposed by Kipple (1964) and described and utilized by Smith and Owensby (1978). Under the IES treatment, yearling steers (~250 kg steer⁻¹) were placed on the grazed site early May (~DOY 128), and were removed late July (~340 kg steer⁻¹; ~DOY 202). The pasture was grazed (0.7-ha per steer for approximately half the growing season, and then the stocker steers were removed in mid July. Soils were silty clay loams (Benfield series: Fine, mixed, mesic Udic Agriustolls). Additional details on the study sites can be found in Bremer et al. (2001), Owensby et al. (2006), and Owensby et al. (2008). The average annual

precipitation at the site was 835 mm and average annual reference crop ET, as computed using the FAO-56 method, was 1470 mm Duisterhaus et al. (2008).

Eddy Covariance and Meteorological Measurements

Eddy covariance (EC) instrumentation for measuring heat, water, and CO₂ fluxes was operated continuously at both sites for three years, 2005 to 2007. Instrumentation included an open-path infrared gas analyzer (LI -7500, LI-COR, Lincoln, NE) to measure the concentrations of H₂O and CO₂ and a sonic anemometer (CSAT3, Campbell Sci. Logan, UT) to measure three dimensional wind velocities, all sampled at 10 Hz. The EC instruments were positioned 3 m above the soil, on relatively flat upland landscapes with over 300 m of fetch in the prevailing wind direction (i.e., south southwest). The distance between towers on the grazed and ungrazed sites was 400 m.

Thirty-min data were available for the entire study for both sites, and post processing of the eddy covariance data included coordinate rotation using the natural wind coordinate, correction for the sonic-derived estimates of sensible heat flux, and density corrections for simultaneous transfer of heat and water vapor. Turbulent fluxes were computed from the 10-Hz time series data using the EdiRe software written by Robert Clement (University of Edinburgh). Corrections included despiking, coordinate rotation, sonic corrections for the sensible heat flux and acoustic air temperature Schotanus et al. (1983), density corrections Webb et al. (1980), and corrections for sensor separation and frequency response. Discussions of the correction procedures can be found in Ham and Heilman (1993) and Lee et al. (2004). Eddy covariance data were filtered to exclude fluxes that failed to meet criteria for wind direction or adequate turbulence. The integral turbulence characteristics (ITC) of the boundary layer were

calculated following the approach of Hammerle et al. (2007). If the ITC test statistic was less than 30% (i.e., well developed turbulence) then the data were accepted. The filtering processes and data loss resulting from inclement weather or sensor failure caused gaps in the dataset. If gaps in the data were less than two hours, and preceded and followed by over two hours of acceptable data, then the gaps were filled by linear interpolation. Otherwise the gaps were filled using diurnal mean method described by Falge et al. (2001) using a seven-day moving window. The fraction of gap filled data ranged from 33 to 38.5 % depending on year and treatment.

One goal of the project was to determine if evapotranspiration from the ungrazed and grazed prairies could be estimated using data from a weather station network. The closest weather station was located at the headquarters of the Konza Prairie Biological Station (KPBS) located approximately 9 km southwest of the prairies. Data from the weather station were used to estimate monthly evapotranspiration using the FAO 56 equation for reference crop evapotranspiration (ET_o) equation Allen et al. (1998).

At the prairie sites, above-ground biomass and leaf area was estimated by harvesting four 0.25 m² areas of the standing vegetation at determined time intervals during the growing season in the footprint of the EC towers at each site. Leaf area was measured from these samples using a leaf area meter (LI-3100, Li-Cor, Lincoln, NE). Biomass was determined from weighing the harvested samples after they have been forced-air oven dried at 55°C for 72 h. This aided in determining effects of grazing by relating the fluxes to the differences in green LAI and biomass for the sites over the season.

Results

Biomass, leaf area index, and climate

Peak biomass on the ungrazed site was similar among years and ranged from 330 to 370 g m⁻² (Fig. 3-1). Grazing reduced peak biomass by 40 to 47 % compared the ungrazed pasture. Owensby et al. (2006) found that regrowth after removal of cattle in July often resulted in a slight increase in biomass in the grazed pastures in August and September. However, in this study results were mixed; some increases in biomass were observed in 2006 and 2007. May and June were periods of maximum growth for both sites. Peaks in biomass usually occurred in August at the ungrazed site, while occurring before August at the grazed site. An exception was during 2006 in which the grazed biomass peak occurred late in September. Green leaf area index (LAI) had slightly higher variation among years, with peak green LAI typically ranging from 2.5 to 3.3 m² m⁻² (Fig 3-2). Ungrazed green LAI peaked in July, and tended to decrease in August and September as the canopy senesced. Grazing caused an average reduction of 47 % in LAI compared with the ungrazed treatment for the study years. Differences in green LAI between sites decreased during August and September as regrowth in grazed vegetation increased. Similar trends were observed by Bremer et al. (2001) and Owensby et al. (2006).

The steers gained approximately 74 kg on average, and gain varied among years from the lowest of 64 kg in 2007, to the highest of 87 kg in 2006 (Table 3-1). Cattle gains seemed to be inversely related to seasonal NCE. The highest gains were reported in 2006, a year that had the lowest biomass and LAI, while 2007 had higher biomass and LAI throughout the grazing season yet had the lowest gains.

Reference evapotranspiration (ET_o) normally peaked in July each year between 180 and 205 mm per month (i.e., 6 to 7 mm d⁻¹) (Fig3-3). Reference ET for the each grazing season (May through October) was approximately 820, 860, and 790 mm for the years 2005, 2006, and 2007, respectively. Monthly total precipitation varied greatly among years (Fig. 3-3). May and June precipitation for 2005 and 2007 was 310 and 330 mm, respectively, while a dry period in May and June 2006 resulted in approximately 70 mm of precipitation. Large differences in ET_o and precipitation also occurred among years, in which May and June precipitation was 10 to 75 mm above ET_o in 2005 and 2007 respectively, while in 2006, May and June precipitation was about 240 mm below ET_o. This likely caused higher levels of water stress and a water deficit in 2006.

Evapotranspiration

Measured evapotranspiration showed seasonal trends among years, with the maximum normally occurring June to July, where ET values ranged from 130 to 150 mm month (4.3 to 5 mm d⁻¹) (Fig. 3-4). Evapotranspiration decreased considerably in September and October due to leaf senescence and decreased ET_o. Evapotranspiration from the ungrazed and grazed sites were very similar, despite the large effect of grazing on the removal of biomass and LAI. Evapotranspiration summed over the growing season (May through October) ranged from 600 to 656 on the ungrazed site and 565 to 632 on the grazed site. Averaged over the grazing season (May to October), ET was reduced by 6, 4, and 3 % in 2005, 2006, and 2007 respectively (Table 3-2).

Figure 3-5 shows that ET from grazed and ungrazed grass prairie was strongly correlated with ET_o and was typically about 25% lower than reference crop ET (i.e., a hypothetical well-

watered grass). This is consistent with the fact the ETo (i.e., evaporative demand of the atmosphere) is typically much larger than precipitation during the summer in the Flint Hills (Fig. 3-3). Figure 3-5 shows that monthly ET can be modeled to within +/- 20 % by simply multiplying ETo by 0.76 for the ungrazed site and 0.74 for the grazed. The differences in the two slopes in (Fig 3-5) confirms that ET from grazed is, on average, about 3 to 4 % lower than on ungrazed prairie. Again, grazing had minimal impact on ET. This suggests that water loss was primarily governed by precipitation and available energy to evaporate water (or weather driven evaporative demand) and was not strongly dependent on LAI. It is likely that transpiration was greater at the ungrazed and soil-surface evaporation was greater at the grazed, but as shown in Figs. 3-4 and 3-5, differences in total ET were small.

There were no strong seasonal patterns in the ET:ETo ratio (Figure 3-6). However, in the prairie the ratio appears to be controlled more by available moisture rather than the phenology and size of the canopy. However, there was a tendency for the ET:ETo ratio to decline to 0.6 or lower as the plants senesced in October. The dependence of the ET:ETo ratio on available moisture suggests that the dual crop coefficient approach (Allen, 2000), which accounts for the effect of specific rainfall events on the ET:ETo ratio, might prove useful when modeling actual ET from tallgrass prairie. Considering the effect of timing between rainfall events would become more important when attempting to predict ET on weekly or daily time scales.

Net Carbon Exchange and NCE:ET

Net carbon exchange (NCE) demonstrated a strong seasonal trend with high rates of carbon exchange in June and July (Figure 3.7). Ungrazed monthly NCE peaked around -175 g C m⁻² (6 g C m⁻² d⁻¹) in June. A rapid reduction in NCE usually started in July as senescence

increased. Grazing reduced the magnitude of NCE in most months; typically causing a 23 to 71 % lower (i.e. less negative) NCE. The biggest difference was in June 2006 when NCE from the grazed and ungrazed sites were -165 and -78.5 g C m⁻² respectively. Late season NCE was not as different as early season, and in some cases, NCE from the grazed site had a higher magnitude than the ungrazed site. This trend was likely due to regrowth and younger leaves, and is similar to the findings of LeCain et al. (2000), Owensby et al. (2006), and Murphy (2007). Net carbon exchange summed over the growing season ranged from -386 to -423 g C m⁻² on the ungrazed and -117 to -404 g C m⁻² on the grazed site (Fig 3-8). The largest difference between the ungrazed and grazed sites was during 2006 (-411 vs. -117 g C m⁻²) when dry conditions in May and June had a strong effect on the photosynthetic capacity at the grazed site (Table 3-2).

Large differences in the NCE:ET ratio were observed between sites. The differences were mainly due to the large differences in NCE (Fig 3-7) and not because of differences in ET (Fig. 3-4). The NCE:ET ratio can be considered as surrogate for water use efficiency (Baldocchi, 1994), and shows that water consumption through transpiration per unit carbon fixed was likely larger at the ungrazed site. Because ET was similar between sites, soil evaporation was likely larger at the more sparse canopied grazed pasture. This is consistent with the conceptual model of Bremer et al. (2001) who outline the effects of grazing on the partitioning of evaporation and transpiration.

Discussion

Biomass accumulation at both sites was greatly enhanced by precipitation during May and June. The precipitation patterns during 2005 and 2007 were more beneficial to biomass

production, unlike 2006 when precipitation was lacking early in the season and later season rain events resulted in limited biomass and green LAI production.

Grazing caused only a 4 % decrease in ET between May and October when averaged over the three years of the study, slightly lower than the 6.1% reduction reported in the single year study of Bremer et al. (2001) and Murphy et al. (2004). The greatest differences in ET (6 %) occurred during 2005, and the least difference was in 2007 at 3 % (Table3-2). There were no strong seasonal patterns in the ET:ET_o ratio (Figure 3-6). This is unlike findings in many cropping systems where the ET:ET_o has a strong seasonal pattern correlated with canopy size. In the prairie, however; the ratio appears to be controlled more by available moisture rather than the phenology and size of the canopy. The ungrazed treatment had higher biomass, which was able to shade the surface from solar loading, unlike the grazed site. This may have lowered soil water evaporation from the ungrazed site while allowing more soil-surface evaporation from the grazed treatment. Lower biomass at the grazed site also may have caused a decrease in transpiration because less leaf area was available to transpire water. Grazing may have also increased aerodynamic resistance to water vapor transport, and lowered ET from the grazed site, similar to Bremer et al. (2001).

Grazing greatly influenced the magnitude of NCE, causing an average decrease among years of 33 % among treatments with the highest difference occurring in 2006 at 72%. Grazing also causes a 33% decrease in the apparent water use efficiency (NCE:ET), with the biggest difference (91 %) occurring in 2006. These differences may have been caused by a decrease in productivity for the year, likely due to decrease in seasonal precipitation. The grazed site had a higher NCE:ET ratio in 2005, which indicated that the younger leaves may have had better apparent water use efficiency. The NCE:ET ratio was higher at ungrazed site for 2006 and 2007.

A similar trend was noticed after cattle removal, during which time NCE:ET on the grazed treatment was often higher than the ungrazed treatment. The main causes in the difference in ET and NCE were likely due to the lack of precipitation during high environmental demand. Biomass and green LAI differences between the sites may also help explain why the differences existed. The ungrazed site had higher biomass and green LAI much of the season allowing for higher NCE rates as compared with the grazed site. However, as the season progressed NCE rates began to merge, and between site differences began to decrease. Interception of light and older plant tissues likely caused the decreased NCE at the ungrazed site, while increased light interception and younger leaf tissues allowed more efficient carbon fluxes at the grazed site, as was also found by LeCain et al. (2000), Suyker and Verma (2001), and Owensby et al. (2006). In 2006, the largest difference in NCE was observed between sites. This was likely the result of decreased precipitation during the early part of the growing season which decreased plant productivity. Bremer et al. (2001) has shown that soil water contents can be lower at this ungrazed site. The higher rates of NCE at the ungrazed site may be explained by the possibility that the site had a larger root biomass compared with the grazed site. Schuster (1964) found that the roots of grazed grasses often had less branching and were sparse than ungrazed grass roots. and Engle et al. (1998) found that grazing can lead to reduced root biomass by reducing root growth, thus limiting the amount of water and nutrient uptake. This may have allowed the ungrazed site to gain water deeper in the soil profile that the grazed site could not attain. Similar findings were observed by Frank (2004) when studying CO₂ flux measurements of grazed and ungrazed prairies. Since ET was similar among sites, the effects of grazing on NCE likely resulted in the large differences in the NCE:ET ratio.

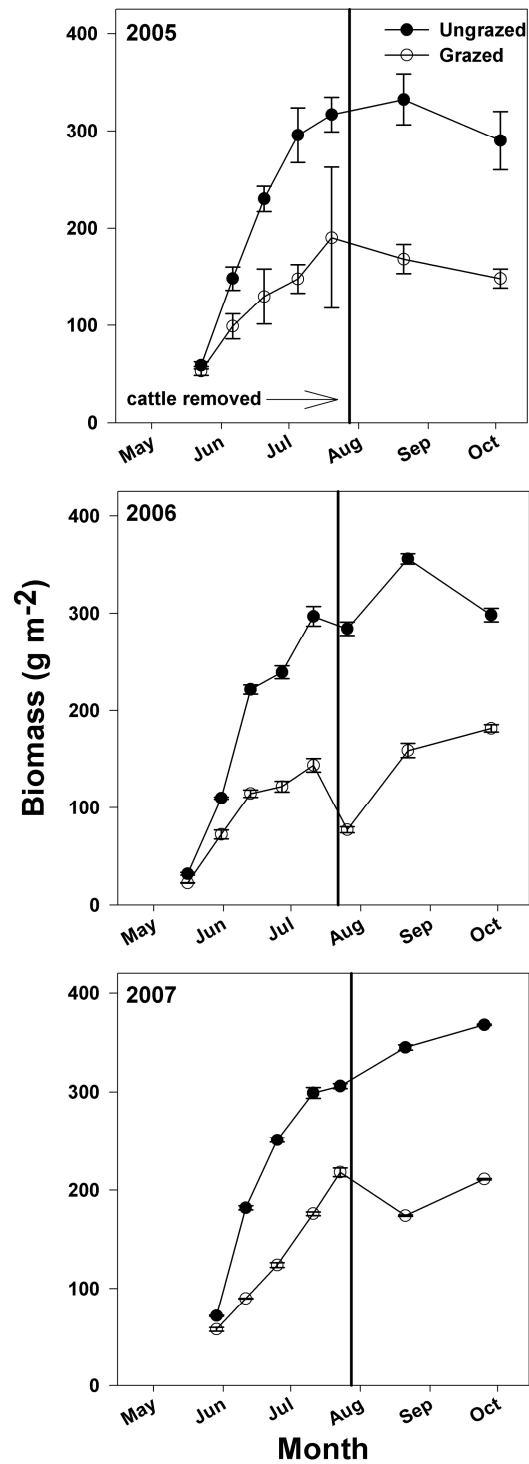
Net carbon exchange should be a measure of ecosystem productivity from a carbon perspective (i.e., total biomass accumulation), yet there was an inverse correlation between cattle weight gain and seasonal NCE (Table 3-1 and 3-2). This demonstrates that there are other factors besides canopy productivity that impact cattle gains (e.g., above and below ground growth, leaf age, nutrient content, and digestibility).

Implications

The goal of this study was to determine if removal of biomass by grazing would affect the ET of the tallgrass prairie. Intensive early grazing reduced ET between May and October by 4% over the three years of the study. The largest difference in ET was observed in 2006, when a 6% ET reduction by grazing was noticed. In general, grazing is not expected to cause a vast impact on surface hydrology on a monthly or longer time scales. Depending on the accuracy needed, models of prairie hydrology may not need to include detailed submodels of herbivory (i.e., stocking density) and dynamic LAI prediction. Monthly ET can be predicted quite well by simply multiplying ETo by 0.76 for the ungrazed site and 0.74 for the grazed. This suggests that a simple reference crop approach for modeling ET from tallgrass prairie might be useful way to make a first approximation of ET from tallgrass prairie. Considering the effect of timing between rainfall events would likely become more important when attempting to predict ET on weekly or daily time scales. Future studies may want to place more emphasis on determining how grazing affects partitioning of ET between transpiration from the plant canopy and evaporation from the soil. The large differences in NCE and NCE:ET ratios between sites showed that grazing has a much stronger effect on carbon fluxes than on ET. However, cattle gains were not correlated with

grazing season NCE. This demonstrates that other factors, such as leaf age, leaf nutrient content, and digestibility, are impacting cattle gains.

Grazing seems to have greater effects on the NCE, and deserves additional research exploring the impacts grazing on the carbon balance of grazed lands.



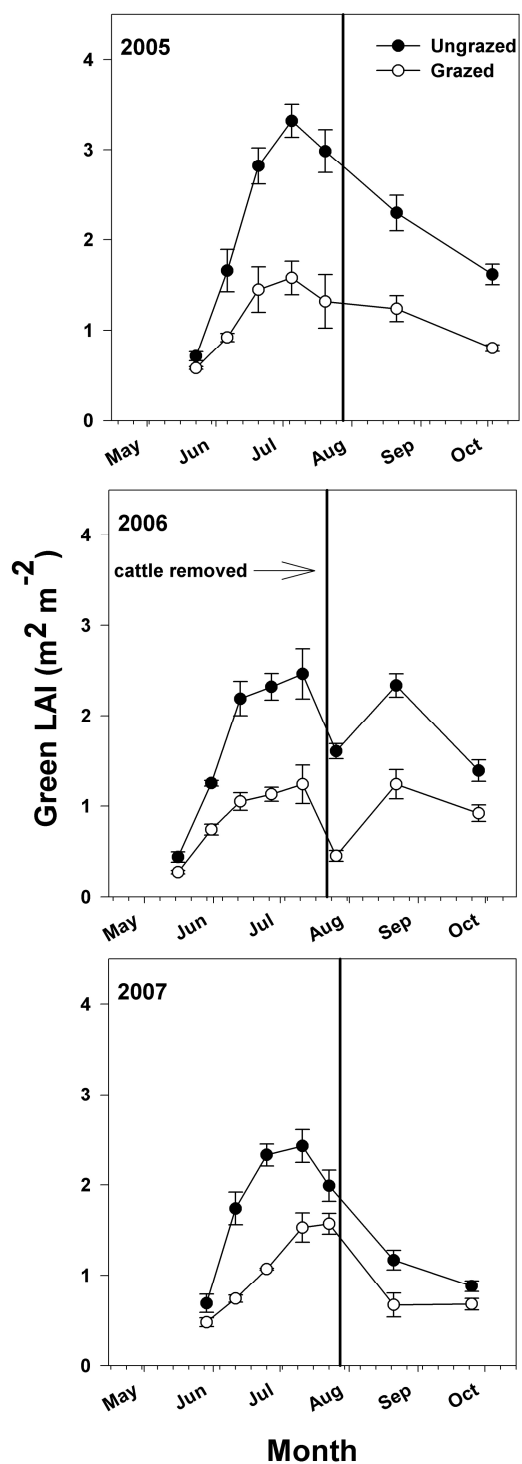


Figure 3-2. Seasonal trends in green leaf area index (LAI) among years compared between the ungrazed and grazed sites (error bars represent the standard error of the sample).

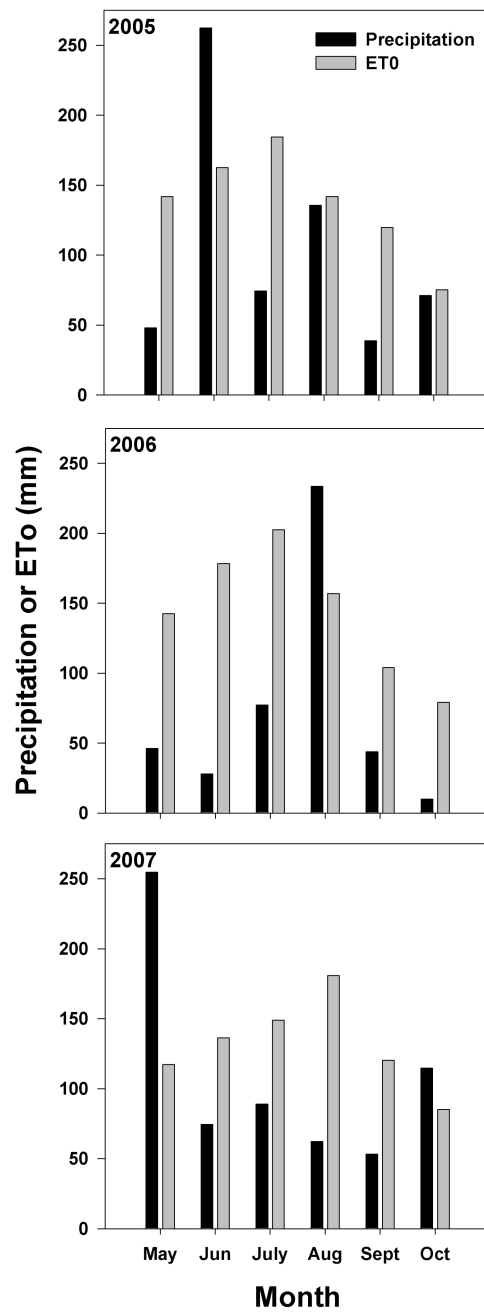


Figure 3-3. Monthly precipitation and reference crop evapotranspiration (ET₀) during the study.

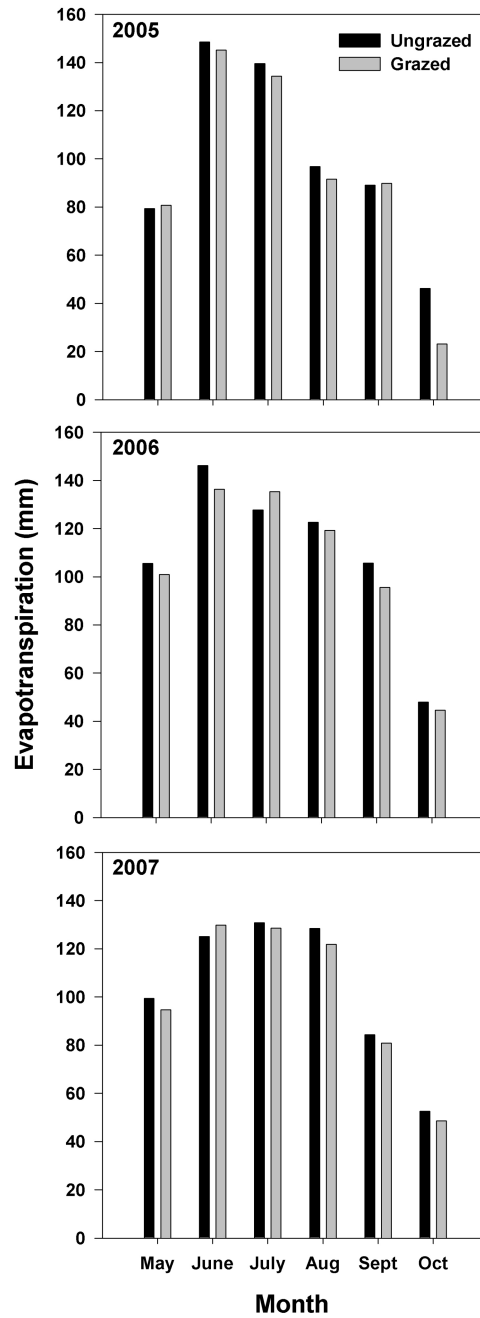


Figure 3-4. Monthly evapotranspiration as measured by the eddy covariance towers for the ungrazed and grazed treatments.

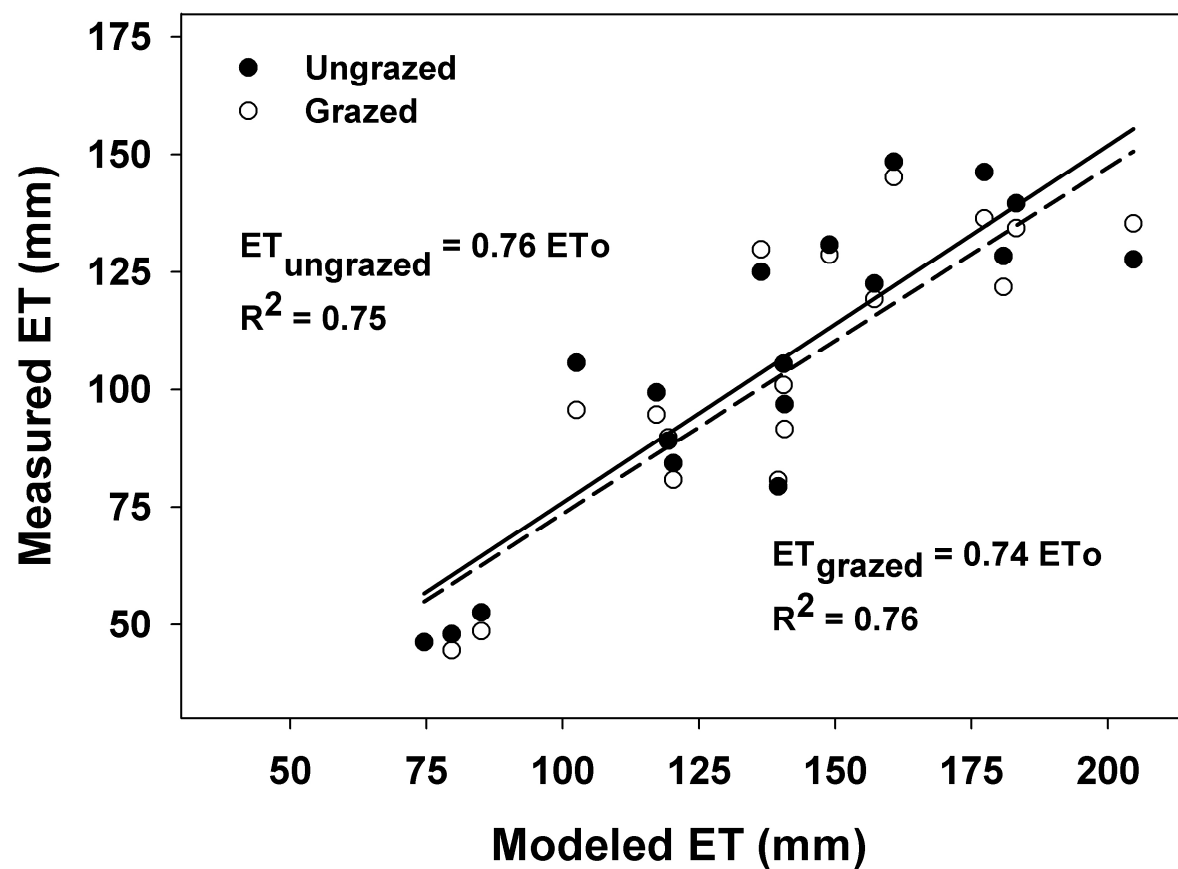


Figure 3-5. Comparison of measured monthly ET for the ungrazed (—) and grazed (---) sites to the FAO-56 reference crop ETo calculated from weather data.

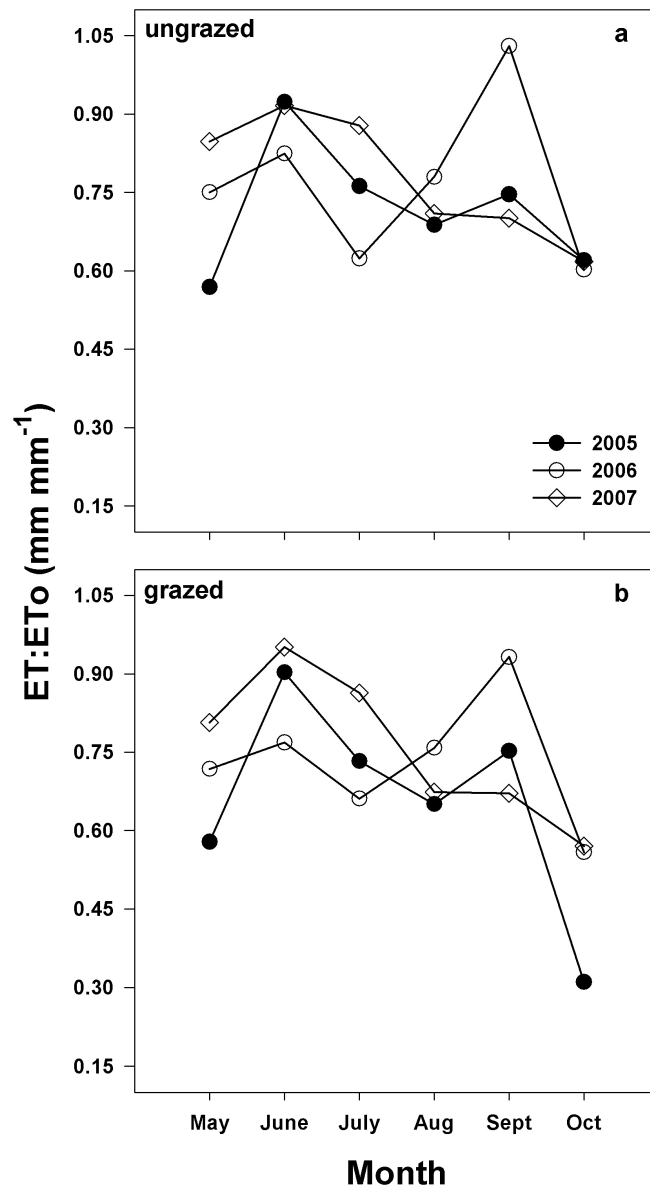


Figure 3-6. Ratio of measured ET to ETo on a monthly basis from the ungrazed (a) and grazed (b) sites to monthly total ETo calculated from weather station data.

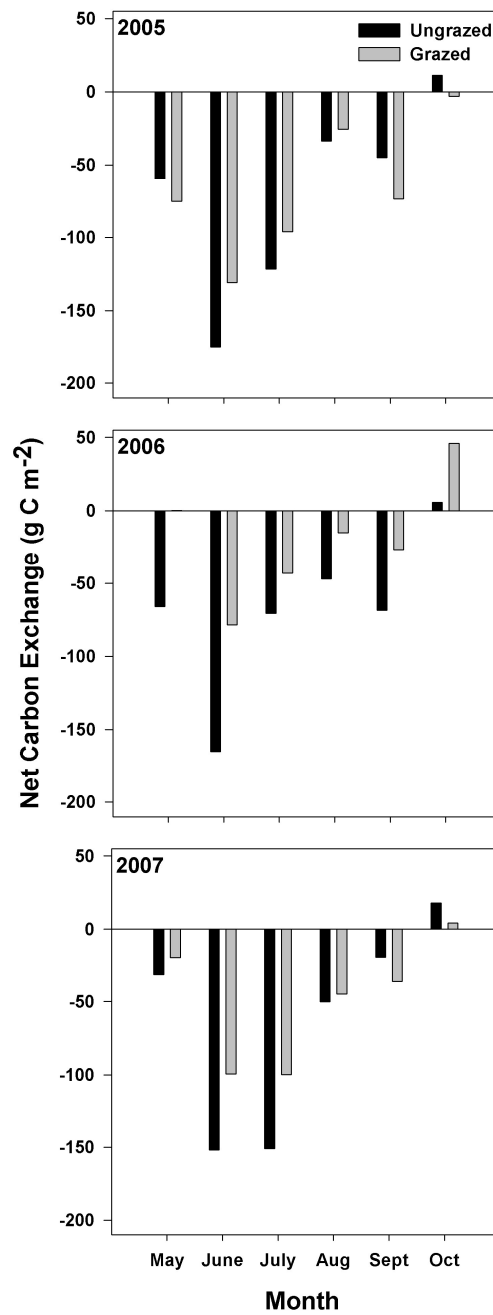


Figure 3-7. Monthly total net carbon exchange (NCE) compared between the ungrazed and grazed treatments. Negative values represent a gain of carbon to the system (i.e., downward net flux).

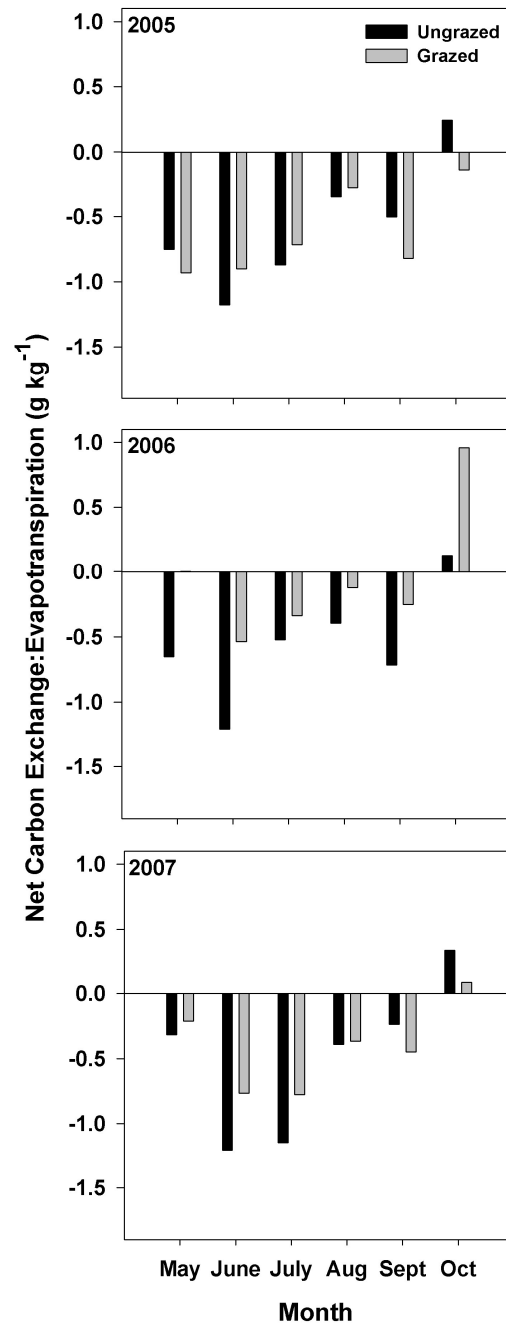


Figure 3-8. The ratio of NCE to ET compared between the ungrazed and grazed treatments. Negative numbers indicate the system was gaining carbon (net downward carbon flux).

Table 3-1. The stocking date, density, initial and ending weights, and the average gain for the cattle put to pasture on intensive early stock grazed pasture.

Year	Stocking Date		Stocking Density	Initial Weight	Ending Weight	Average Gain
	Placed on	Removed	ha steer ⁻¹	_____	kg steer ⁻¹	_____
2005	5/14/2005	7/28/2005	0.81	274	344	70
2006	5/2/2006	7/22/2006	0.81	221	307	87
2007	5/14/2007	7/28/2007	0.81	209	273	64

Table 3-2. The yearly total evapotranspiration (ET), net carbon exchange (NCE), and the ratio of NCE to ET compared between the ungrazed and grazed sites. Negative numbers represent a sink (or gain) of carbon to the system.

Year	ET (mm)*		NCE (g C m ⁻²)		NCE:ET (g kg ⁻¹)	
	Ungrazed	Grazed	Ungrazed	Grazed	Ungrazed	Grazed
2005	600	565	-423.0	-404.3	-0.57	-0.63
2006	656	632	-411.9	-117.3	-0.56	-0.05
2007	620	604	-386.4	-295.9	-0.50	-0.41
Average	625	600	-407.1	-272.5	-0.54	-0.36

* a mm of evaporation is equal to a kg of water per m².

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CHAPTER 4 - Conversion of Tallgrass Prairie to *Juniperus virginiana*: Effects on Rainfall Interception, Evapotranspiration, and Net Carbon Exchange

Introduction

Prescribed burning of tallgrass prairies in the Flint Hills of Kansas is necessary to sustain the ecology and economic productivity of this unique native ecosystem. Over a million head of cattle are grazed on the Flint Hills each summer. Many pastures are burned each spring to remove the accumulated grass residue and mulch and to promote vigorous regrowth. Furthermore, burning helps prevent woodland encroachment by killing young trees and shrubs. Burning is often completed in April, several weeks before stocker cattle fill the prairies. Unfortunately, in certain areas of Northeast Kansas, the dominant vegetation is changing rapidly from grassland into woodland. Briggs et al. (2002) found a 5.8% yearly expansion rate for an eastern red cedar (*Juniperus virginiana*) stand. Knapp et al. (2008) summarized work by Hoch et al. (2002) and Heisler et al. (2003) reporting a tallgrass prairie ecosystem can be converted to a closed-canopy eastern red cedar forest in as little as 40 years. Owensby et al. (1973) found that routine burning of the prairie was one of the best methods to remove eastern red cedars from the landscape, and works especially well on eastern red cedars less than 1.8 m in height. Grazing and burning, when managed properly, will not only improve range conditions, but can also increase primary production and cattle weight gains. Owensby (2005) stated that as these management

practices become relaxed, many range areas have changed in botanical composition. The loss of prairie to woody encroachment has an immediate economic impact by reducing the amount and productivity of area that can be used for grazing. As eastern red cedar encroachment expands, rainfall reaching the soil surface may decrease dramatically due to canopy interception. Water caught by the canopy readily evaporates, reduces runoff and deep percolation as well as water available for soil evaporation and plant transpiration. Thus, the hydrology of the region could be altered by woodland expansion. Gash and Stewart (1977) reported 35 % year-end average interception of bulk rainfall in a mixed stand of Scots (*Pinus sylvestris* L.) and Corsican (*Pinus nigra* var. *maritima* (Ait.) Melv.) pine. Other canopy interception research, such as Llorens et al. (1997), has shown a 24 % interception of bulk rainfall in a *Pinus sylvestris* stand, Carlyle-Moses (2004) and Asdak et al. (1998) found that interception could be quite low and account for only 8.2 to 11 % of the bulk rainfall, respectively. Owens et al. (2006) observed a 35 % interception of bulk rainfall by juniper woodlands, and Thurow and Hester (1997) showed that as juniper cover increased from 0.0 to 36 %, precipitation reaching the soil was reduced by 34 %. They conclude that any pasture vegetation conversion from brush to grass will increase soil available water. Stewart (1977) and Singh and Szeicz (1979) found that evaporation of intercepted water was two to three times higher than transpiration, and Pearce et al. (1980) estimated evaporation of intercepted water at a rate of 0.37 mm hr⁻¹. Fleischbein et al. (2005) and Guevara –Escobar et al. 2007 found that interception rates can be as high as 50 % of the received precipitation. Grassland canopy interception is often lower than woodland interception, and has been shown to be around 5 % of the total yearly precipitation Corbett and Crouse (1968) to 19 % Gilliam et al (1987).

Interception of precipitation by the woodland canopy could cause a difference in the energy partitioning and microclimate of the woodland ecosystem as compared with the grassland ecosystem. The increasing cedar populations are expected to decrease the fluxes of latent heat and soil heat, while increasing the sensible heat flux. Higher Bowen ratios (i.e., the ratio of sensible and latent heat) are expected from closed-canopy woodland areas based upon physiological differences controlling stomatal conductance of water vapor. While studying a coniferous forest, Lindroth (1985) found that 59 % of the net radiation was used for latent heat flux, and 32 % was used for sensible heat flux. Of this percentage only 13 % of latent heat flux came from the ground vegetation, while 50 % of the sensible heat flux came from the ground. Kelliher et al. (1989) and Baldocchi and Vogel (1996) found that latent heat flux above a *Pinus radiata* D. Don canopy was strongly determined by stomatal conductance, vapor pressure deficit, and leaf area index. Kelliher et al. (1993) found that the relationships of surface water conductance and atmospheric saturation deficit were more similar above a coniferous canopy than above a grassland canopy. Baldocchi and Vogel (1996) and McNaughton and Jarvis (1983) explain this trend, and discuss that grassland transpiration is controlled by net radiation, whereas saturation deficit controls forest canopy transpiration. Grossnickle et al. (2005) found that as the vapor pressure deficit increased, net photosynthesis and canopy water conductance decreased. This led to a lower water use efficiency among the western red cedar (*Thuja plicata*) populations.

Conversion of grasslands to woodlands also affects the other components of the biogeochemical cycles McKinley et al. (2008). Norris et al. (2001) found that annual aboveground net primary productivity ranged from 7 250 to 10 440 kg ha⁻¹ year⁻¹ compared to only 3 690 kg ha⁻¹ year⁻¹ for tallgrass prairies sites, and Smith and Johnson (2004) found little

change in soil organic carbon between prairie and forested sites suggesting only small changes in carbon storage. They conclude that eastern red cedar expansion may have consequences to carbon cycling in the region. Soil water is expected to be lower at the woodland site Smith and Johnson (2004) and McKinley et al. (2008) compared with the grassland sites due to canopy interception of precipitation. Knapp et al. (2008) reported that summertime leaf scale photosynthetic activity was often three times lower in eastern red cedar when compared with the native C4 grass *Andropogon gerardii*.

Changes in the energy and water balance caused by woodland expansion could alter regional scale hydrology and mesoscale weather patterns Avissar and Pielke, (1991) and Lynn et al., (1995). Therefore, patchy woodland establishment could ultimately affect the productivity of grasslands over large regions by altering patterns of precipitation.

The goal of this research was to compare water, carbon, and energy fluxes of a native tallgrass prairie to those from a closed-canopy stand of eastern red cedar that had replaced a tallgrass prairie. Long-term eddy covariance measurements at both prairie and woodland sites were compared over a one-year period in the Kansas Flint Hills. Also included were studies of rainfall interception, stemflow, and through fall at the forested site. These data coupled with ancillary soil and plant measurements are used to show how conversion of prairie to woodland can ultimately affect the hydrology in land areas undergoing this conversion.

Methods

Study sites included an ungrazed tallgrass prairie and a stand of eastern red cedar woodland located in the Flint Hills of northeastern, Kansas USA. The prairie site was part of the Konza Biological Station (watershed 1D) approximately 5 km south of Manhattan, KS (lat 38° 08'N, long 96° 32'W, and ~385 m above mean sea level). The vegetation was dominated by native C₄ grasses, including big bluestem (*Andropogon gerardii* Vitman) and indian grass (*Sorghastrum nutans* (L.) Nash). The prairie site was annually burned each spring and ungrazed for the last several decades. Soils were silty clay loams (Benfield series: Fine, mixed, mesic Udic Agriustolls). The woodland site was an approximately 45-yr-old stand of eastern red cedar (*Juniperus virginiana*) (C₃ metabolism) located 40 km north of Manhattan, immediately north of Randolph, Kansas (lat 39° 43'N, long 96° 76'W, and ~325 m above mean sea level). The site had once been tallgrass prairie, but burning and grazing ceased in the 1950s and allowed the cedar invasion. Tallgrass prairie still bordered the site to the south. The woodland was a closed canopy with no understory vegetation. Norris et al. (2001) estimated total above ground biomass at the site was 120 739 kg/ha with a density of 1900 trees/ha. The average tree height was 9 m and the average diameter was 0.11 m. Soils were silty clay loams (Irwin series: Fine, mixed, mesic Pachic Agriustolls). Additional details on the study sites can be found in Norris et al. (2001), and Smith and Johnson (2004). Yearly precipitation is approximately 835mm at the prairie site, 780mm at the woodland site. Precipitation occurring during the months April through October accounts for 80% of the total for both locations.

Eddy Covariance and Meteorological Measurements

Eddy covariance (EC) instrumentation for measuring heat, water, and CO₂ fluxes was operated continuously at both sites between May 2004 to April 2005. Instrumentation included an open-path infrared gas analyzer (LI-7500, LI-COR, Lincoln, NE) to measure the concentrations of H₂O and CO₂ and a sonic anemometer (CSAT3, Campbell Sci. Logan, UT) to measure three dimensional wind velocities, all sampled at 10 Hz. The EC instruments were positioned at 14 m at the woodland site and 3 m at the prairie site. The prairie tower was located on a relatively flat upland landscape with over 400 m of fetch in the prevailing wind direction (i.e., south-southwest). The woodland site had 300 m of fetch in the prevailing wind direction and was bordered to the south by tallgrass prairie. While more upwind fetch at the woodland was desirable, it was challenging to find a mature woodland site with uniform vegetation. Also, internal boundary layers develop quickly over forests when a short smooth surface is upwind (i.e., prairie).

Energy balance instruments included a net radiometer (Q9, Radiation and Energy Balance Systems., Seattle, WA) and three soil heat flux plates (HFT-3, Radiation and Energy Balance Systems).

Air temperature and relative humidity were measured at the height of the EC instruments at both sites using a HMP35A probe (Vaisala, Helsinki, Finland) and precipitation was measured with a tipping bucket rain gauges (TE-525 Campbell Sci, Logan, UT, and a Sierra-Misco model 2501, Nova Lynx Corp., Grass Valley, CA). The 10-Hz time series data from the open path analyzers and sonic anemometers were collected on laptop computers.

Turbulent fluxes were computed from the 10-Hz time series data using the EdiRe software written by Robert Clement (University of Edinburgh). Corrections included despiking,

coordinate rotation, sonic corrections for the sensible heat flux and acoustic air temperature Schotanus et al. (1983), density corrections Web et al. (1980), and corrections for sensor separation and frequency response. Discussions of the correction procedures can be found in Ham and Heilman (1993) and Lee et al. (2004). Eddy covariance data were filtered to exclude fluxes that failed to meet criteria for wind direction, fetch, or adequate turbulence. The integral turbulence characteristics (ITC) of the boundary layer were calculated following the approach of Hammerle et al. (2007). If the ITC test statistic was less than 30% (i.e., well developed turbulence) then the data were accepted. The filtering processes and data loss resulting from inclement weather or sensor failure caused gaps in the dataset. The gaps in the time series (approximately 35 % of the total) were filled using diurnal mean method described by Falge et al. (2001) using a seven-day or nine-day moving window. Net carbon exchange in this text is represented with negative sign if carbon is being gained by the vegetation from the atmosphere (carbon sink) and positive if carbon is being lost to the atmosphere (carbon source).

Measurements of Rainfall Interception

Instrumentation was installed at the woodland site to quantify rainfall interception. Interception (I) was estimated by comparing precipitation (P) measured above the canopy to throughfall (Tf) and stemflow (Sf) collected catch basins positioned below the canopy as,

$$I = P - Sf - Tf \quad (1)$$

Three catch basins were placed in the footprint of an eddy covariance tower approximately 20 m apart (Fig.4-1). Each trough was constructed from a (1.22 x 2.44 m) sheet of 0.85 mm galvanized metal. The metal was then folded length wise to form a V- shape, 0.48-m deep channel, designed after the troughs used by Llorens et al. (1997). The troughs were set atop support stands and positioned approximately one meter above the soil surface, but under the

lowest vegetative juniper branch. Each trough drained into two interconnected 19-L buckets giving a total capacity of 38 L. Depth of water in the buckets was measured automatically by a linear displacement transducer (LX-PA-25, Unimeasure, Corvallis, OR), connected to a float. The depth recorder was similar to that used by Ham and DeSutter (1999). A data logger (HOBO H8 data logger, Onset Computer Corp., Pocasset, MA.) was used to measure the transducer's voltage and store the bucket water depth measurements on 30-min intervals. Each trough had its own data logger and transducer, both housed in a weatherproof enclosure on the bucket lid. The buckets were drained manually after each significant rainfall event. Stem flow (i.e., water running by gravity flow downward along the periphery of the trunk) was measured by wrapping pieces of split 16-mm diameter rubber hose around 5 randomly chosen cedar trees in the vicinity of the throughfall troughs (Fig.4-2). Each piece of hose was first split in half lengthwise and then fastened to each tree in a downward spiral and sealed to the trunk with silicone adhesive. A barb fitting then connected each hose to a piece of plastic tubing that allowed any water to drain into a 11 L reservoir. Collected water was weighed and then converted into millimeters of depth based on tree density. A tipping bucket rain gauge mounted on a 15 m-tower was used to record above-canopy rainfall.

Soil and Plant Measurements

To measure soil moisture, a JMC soil-sampling tube (Clements Associates Inc. Newton, IA) was used to take 5 soil cores at a 0 to 10 cm depth every two weeks along 40-m transects at each site during the growing seasons of 2003 and 2004. Soil gravimetric water contents were then integrated through this depth. Bulk density at each site (1.13 g cm^{-3} prairie to 1.25 g cm^{-3} woodland) was used to convert gravimetric to volumetric contents.

Results

Soil Water Data

Soil water content near the surface was lower at the woodland site in 2003 and 2004 despite similar pattern of precipitation between the two sites (Fig.4-3). Bremer et al. (2000), Smith and Johnson (2004), and McKinley et al. (2008) observed similar trends at and near this site. Soil water content was sometimes 10 to 15 % less at the forest site in the spring of 2003, but differences between sites became less as the soils in each site dried later in the summer. Precipitation events in late August caused water content to increase at both sites, but woodland water contents dropped while remaining high at the prairie. In 2004, water contents started off similar between sites, but began to separate during mid season (early June through mid August), again with water contents 5 to 15 % lower at the woodland. Soil water contents merged in October as the soils at each location dried. On average, the woodland soil water content from 0 to 10 cm was approximately 20 percent lower than the prairie. Given there was almost no understory vegetation at the woodland (i.e., for transpiration) and minimal energy at the soil surface for ET (i.e., effect of canopy shading), the lower soil water content at the woodland was likely caused by interception of rainfall by the *juniperous* canopy.

Rainfall interception Study

Total precipitation at the woodland site during the 10-month interception study (October 27, 2004 to August 26, 2005) was around 750 mm, which was 90 mm above normal. The prairie site received 740 mm, which was 40 mm higher than normal for the same time period. The fraction of precipitation measured as throughfall and stem flow was highly variable

during the water balance study (Fig. 4-4). On average, throughfall was 34%, with an average of 6 mm per event but ranged from 1 to 8 mm. Throughfall amounts were positively correlated to the amount of precipitation (Fig.4-5) and there was not clear evidence of lower fractional throughfall (i.e., greater fractional interception) during small rainfall events. Throughfall amounts greatly increased as precipitation increased from 5 to 15 mm, but only slight increases were observed from 15 to 30 mm. For each rain event, throughfall amounts measured at the three different locations under the canopy were remarkably consistent (Table 4-1). This suggests that a large number of sample locations are not required when using collectors with large sample areas. Stemflow was small, contributing only 14 % of the total rainfall; an amount equal to 1.4 mm on average with a range of 0.1 to 3.6 mm per event. Stemflow measurements among the different collector sites were typically in very good agreement (Table 4-2). Interception of precipitation accounted for 52 % of the total precipitation, but ranged from 17 to 77 % among rainfall events. The dense, mostly horizontal architecture of the mature *juniperous* canopy was very effective in trapping rainfall. Percent interception was inversely related to precipitation rate (Fig.4-6). Precipitation rate explained 74 % of the variation in interception. The majority of the precipitation events had rates under about 3 mm hr⁻¹ resulting in high amounts of water being trapped by the canopy. Precipitation rates over 6 mm hr⁻¹ showed lower interception percentages resulting in increased amounts penetrating the canopy. The measured interception was higher than the 24 %, 30 % and 42 % average interception given by Llorens et al. (1997), Gash and Stewart (1977), and Thurow and Hester (1997), respectively, yet were very similar to the findings of Fleischbein et al. (2005) studying *Ficus*, *Hyeronima*, and *Piper* trees in lower montane rainforests and Guevara-Escobar et al. (2007) studying *Ficus* trees in urban landscapes.

Prairie vs. Woodland Evapotranspiration

Despite the large differences in the plant canopies, differences in ET between the grassland and woodland were small throughout the year, (Fig 4-7). Evapotranspiration for both sites peaked in July, and showed to be equally distributed throughout the summer of 2004. Total annual ET for the prairie was around 813 mm, with a monthly range of 12 mm for December (0.4 mm d^{-1}) to a high of 156 mm for July (5.2 mm d^{-1}); Total annual ET from the woodland site was 885 mm with a monthly range of 12 mm in January (0.4 mm d^{-1}) to 183 in July (6.1 mm d^{-1}). During the active part of the growing season for the grassland (May to September), monthly ET from the prairie was typically greater than from the woodland (Fig. 4-7). An exception was in July, when woodland ET was nearly 30 mm higher than the prairie. This may have been the results of evaporation of intercepted rain directly from the woodland canopy. Monthly ET rates for the woodland site were higher in the spring and fall when the evergreen canopy allowed for transpiration but the prairie canopy had senesced. Annual total ET for the woodland site 885 mm compared to 813 mm from the prairie site, a difference of 8 %. This was similar to Dugas and Mayeux (1991), who found a 7 % decrease in seasonal evaporation from grassland after defoliation of mesquite trees. The mechanisms affecting ET from both sites will be evaluated in more detail in the energy balance section.

Net carbon exchange

While ET among sites was similar throughout the year, large differences were observed in NCE (Fig. 4-8). The prairie demonstrated a seasonal NCE pattern typical of perennial vegetation in a temperate climate, showing large carbon movement into the ecosystem in May through August; the rapid growth phase following the prescribed burn. These patterns are

similar to those of Suyker et al. (2003) and Owensby et al. (2006) who made flux measurements on the same prairie ecosystem. Conversely, the woodland showed much lower NCE during the spring and summer, and in stark contrast to the prairie, very minimal seasonal variation. Monthly NCE at the prairie between May and August was -130 to -170 g C m^{-2} (-4.3 to $-5.7 \text{ g C m}^{-2} \text{ d}^{-1}$), while carbon gains at the forest over the same period were three to four times smaller. The woodland became a net source of carbon in July. High temperatures likely reduced photosynthetic capacity of the *Juniperus* canopy while soil and bole respirations may have increased. Conversely, the greatest carbon gains in the woodland occurred during March through April when there was adequate moisture but lower air temperatures. Higher ecosystem respiration may have reduced the NCE during July, 2004. Since precipitation was above normal for the month, it is likely that the soil and surface litter may have been wetter than the other months of the study. Wetter soils and litter layers tend to have higher amounts of microbial decomposition and break down of carbon compounds, resulting in higher amounts of respired carbon. Similar trends in woodland soil respiration were observed by Davidson et al. (1998) and Rey et al (2002). Different patterns in NCE between the sites were expected due the differences in plant physiology and phenology between the eastern red cedar and prairie grasses; the prairie was governed by a perennial growth pattern and water limitations while the evergreen woodland was more strongly affected by temperature. Patterns of NCE were consistent with the leaf level gas exchange measurements of eastern red cedar and prairie grasses reported by Knapp et al. (2008). Over the year, total carbon flux from the prairie was -506 g m^{-2} while only -264 g m^{-2} from the woodland (Table 4-3). However, the prairie lost approximately 218 g m^{-2} of carbon during the prescribed burn in the spring of 2005, reducing total carbon flux to -288 g m^{-2} ; thus, depending on the period of integration, the annual carbon balance of the prairie was close to

zero. Others have shown the carbon balance of tallgrass prairie to be close to equilibrium Sukyer et al. (2003) and Owensby et al. (2006).

Differences in NCE between the two sites resulted in large differences in the NCE:ET ratio (Fig. 4-9). Here the NCE:ET ratio is considered a surrogate for water use efficiency (WUE), or a general measure of how relationships between carbon and water fluxes vary between systems and over time. The NCE:ET ratio generally followed the same trend as NCE. The most negative NCE:ET ratios (largest apparent WUE) from the prairie occurred during May to September. Carbon gains per unit of ET were approximately four times greater during the study than were measured from the woodland site during the summer months. The NCE:ET ratio improved for the woodland communities during the early fall of 2004 (Table 4-3).

Prairie vs. Woodland Energy Balance

There were only small differences in ET between the prairie and woodland, but this does not mean conversion may not have affected other aspects of the energy balance. A detail study was made of the midday energy balance in August, 2004; a period of dry weather that followed above average rainfall in July. Net radiation (R_n) over the month of August 2004 was approximately 100 W m^{-2} larger at the woodland site than at the prairie site (Fig 4-10). Because there were no differences in global irradiance between the sites, the differences in R_n were likely caused by lower albedo at the woodland. This means that the woodland site had higher available radiant energy that could be used to evaporate water, heat the air, or store heat in the soil and canopy. Over the same period, the average daily sensible heat flux (H) from the prairie site was 87 Wm^{-2} , about 2.5 times lower than the woodland sites flux of 215 Wm^{-2} (Fig. 4-11a). However, daily latent heat fluxes (LE) were within 10 % between locations, where the average

values where 304 to 275 W m⁻² at the prairie and woodland locations respectfully (Fig.4-11b). The large differences in H between the sites caused difference in the Bowen Ratio (BR), expressed as the ratio of H:LE (Fig. 4-12). On average, the BR from the prairie site was 26% lower than that observed at the woodland.

Discussion

Of the total precipitation, 54% was intercepted by the canopy and likely lost rapidly by direct evaporation. This may help explain the differences in soil water content for the woodland site, and why the site is consistently lower than the prairie. Interception of rainfall may also help explain why ET in July was higher at the woodland site even though small differences in precipitation were recorded. Water from rain events likely evaporated from the woodland canopy, while the water at the prairie likely went to runoff, deep percolation, or soil storage. This may also explain why the soil water content was much lower at the woodland site during this time period (Fig 4-3). Canopy interception was the greatest during winter and early spring when precipitation events were small and less intense than summer precipitation events. Some of the variability in throughfall was explained by the amount and rate of precipitation falling to the canopy (Figs.4-5 and 4-6). The precipitation events that occurred during 27 Oct.to 19 Nov.,, 2004 were typical of the normal fall precipitation events, and consisted of smaller amounts and less intensive precipitation. In theses events, the woodland canopy intercepted much of the precipitation 23 mm, to only 16 mm of throughfall. Interception continued until the canopy became fully saturated and can no longer hold water. These events are different that the early spring and summer events of 2005, in which precipitation duration and intensity were event dependent. The events of 20 July and 26 Aug. were comprised of very short duration and intense precipitation rates, which lead to higher throughfall (13 mm) compared with interception (6 mm). However, not all variability in canopy throughfall can be based on amount and rate of

precipitation. Environmental factors such as seasonal precipitation trends, winds speed during precipitation event and pre-precipitation conditions may all control the partitioning of precipitation in the woodland canopy. Winter precipitation accounted for only 20% of the total precipitation for the year, and was characterized by long duration low intensity events than the spring and summer events.

Even though evapotranspiration was similar between the sites during the study, net carbon exchange was different causing differences in the NCE:ET ratio. Carbon fluxes matched the normal growth patterns typical of the C₄ grasses, with peaks in NCE occurring from May to June, and slowly tapering off as plant age increases. Different patterns in NCE were observed from the woodland site, and were expected due the difference in plant physiology and phenology of the eastern red cedar. The eastern red cedar has a more indeterminate and continuous growth pattern allowing it to undergo photosynthesis throughout the year. This allowed for steady, yet sometimes lower carbon gains throughout the year as compared with the native prairie. The phenology of the prairie grasses only allows for approximately 5 months of photosynthetic activity resulting in a narrower time frame for carbon gains. These differences became less apparent into the cooler fall months as the cedar populations tended to become more of a sink of carbon than a source. These results are similar to Knapp et al. (2008) and support that the grass prairies are more adept at fixing carbon during the hotter summer months than are the cedar woodland communities, yet are limited by soil available water. Each site showed a year end net gain of carbon, which may have been a result of above normal precipitation at the prairie (+ 40 mm) and at the woodland (+20 mm).

The woodland likely had higher amounts of stored energy in the canopy causing the sensible heat flux to be greater at the site. Several possible reasons can explain this occurrence

including lower albedo and decreased soil heat storage at the woodland site. Forested lands often have a much lower albedo (e.g., 0.15) compared with grasslands (e.g., 0.26) Campbell and Norman (1998). This surface characteristic leads to increased energy being absorbed by the woodland canopy. This was observed during the middle of August, 2004 (Fig.4-10) in which the net radiation (R_n) was approximately 100 W m^{-2} higher at the woodland site. Because the latent heat fluxes were the same, more of the available energy went into sensible heat, and increased the Bowen ration at the woodland site. Bowen ratios in both sites started off low, but increased during the month, similar to the H patterns. Since the woodland canopy intercepted more of the solar radiation, less energy went into soil storage, which also helps explain the rise in sensible heat flux. The thicker woodland canopy may have absorbed more energy as compared with the prairie site, leaving less to be partitioned into the soil.

Implications

The goal of this research was to show how conversion of prairie to woodland can ultimately affect the hydrology in land areas undergoing woodland conversion. Results from this study indicate that forestation may impact the surface water balance of the tallgrass prairie. Conversion of the tallgrass prairie into cedar woodlands can limit production not only by decreasing available land for grazing, but by limiting the available surface water from interception of precipitation. Cedar encroached areas can intercept over half of the received precipitation, which may have impacts on infiltration and surface water flows. As cedar forestation increases, less water from precipitation will reach the soil surface. This will lead to diminished surface water flows from forested areas. As surface water flows decrease, the amount

of water that is channeled into stock watering ponds and reservoirs will decrease, causing shortages in available drinking water for grazing livestock.

Woodland conversion increases available radiant energy by changing the surface reflectance and decreasing albedo. Campbell and Norman, (1998), Betts and Ball (1997), and Giambelluca et al. (1997).

Woodland conversion had little effect on the overall ET compared with prairie. The increased interception likely decreases soil water content, which limits understory growth of other vegetation. The similarities in ET between sites may be caused by a difference in partitioning of ET. It is likely that the woodland site has increased evaporation from intercepted precipitation, where as the prairie site has higher transpiration since more of the water from precipitation can be conducted to the soil. Since available solar radiation is higher in the woodlands compared with the prairie, and LE is similar, more energy is converted into H and the BR is increased. Increased heating over the cedar woodlands impact regional climate, including precipitation patterns form the woodland areas as discussed by Henderson-Sellers et al. (1993) and Pielke et al.(1998).

As forestation increases, canopy interception of rainfall may be detrimental to the surface water flow and productivity of the tallgrass prairie, and therefore is important to limit Eastern red cedar forestation of the tall grass prairie.

Acknowledgements

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Figure 4-1. Photograph of a throughfall collector (area: 0.30 x 2.44 m) positioned below the woodland canopy.



Figure 4-2. Photograph of a stemflow collector attached to a tree (total length: 735mm).

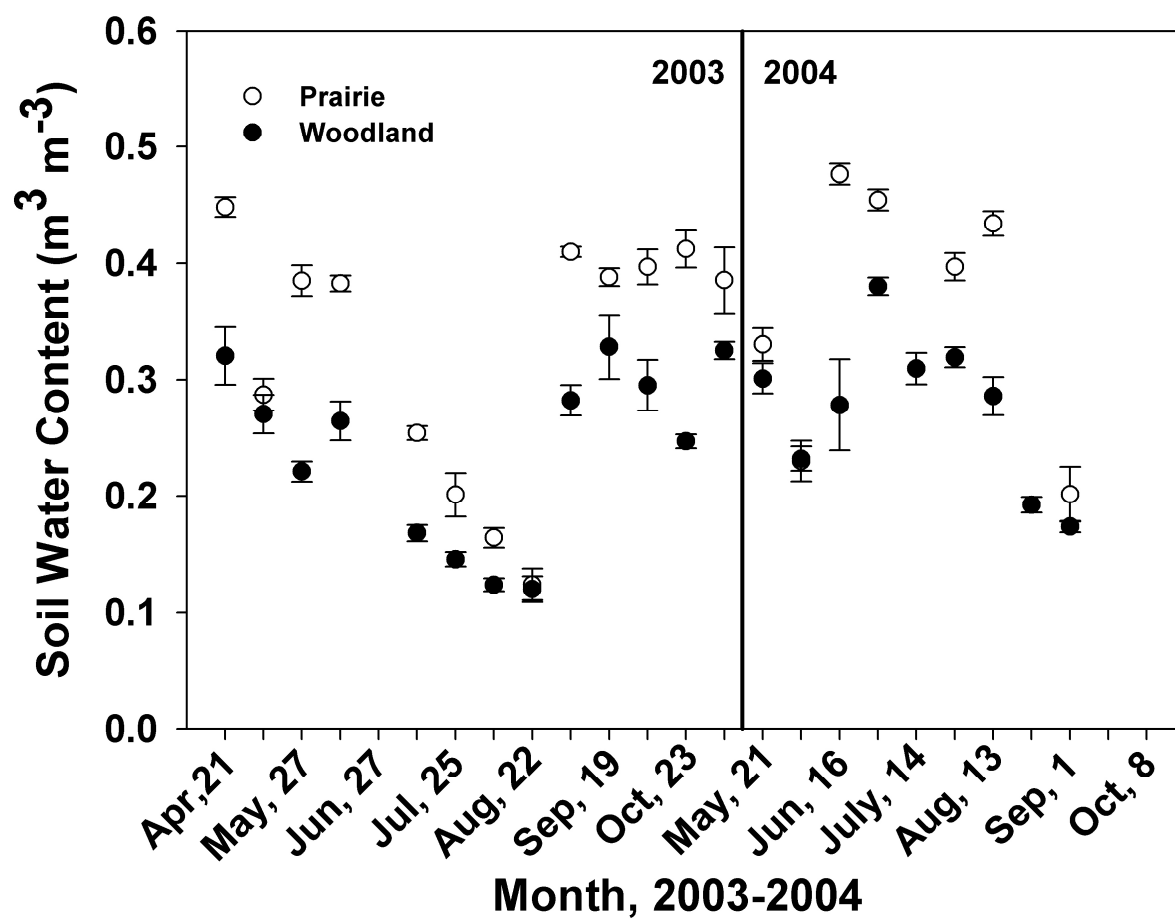


Figure 4-3. Volumetric soil water content (0 to 10cm) at the prairie and woodland sites.

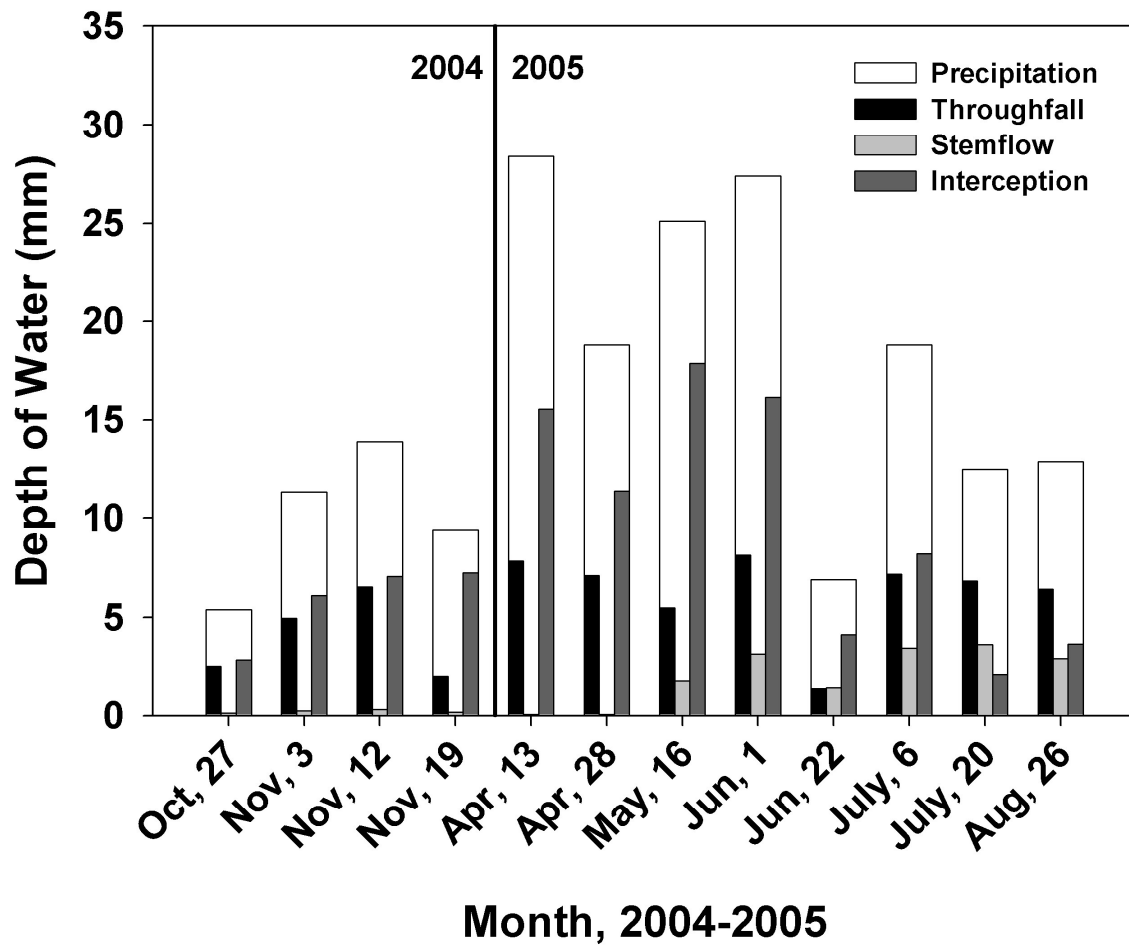


Figure 4-4. Measurements of throughfall, stemflow, and precipitation at the woodland site. Also included is interception as calculated from equation 1.

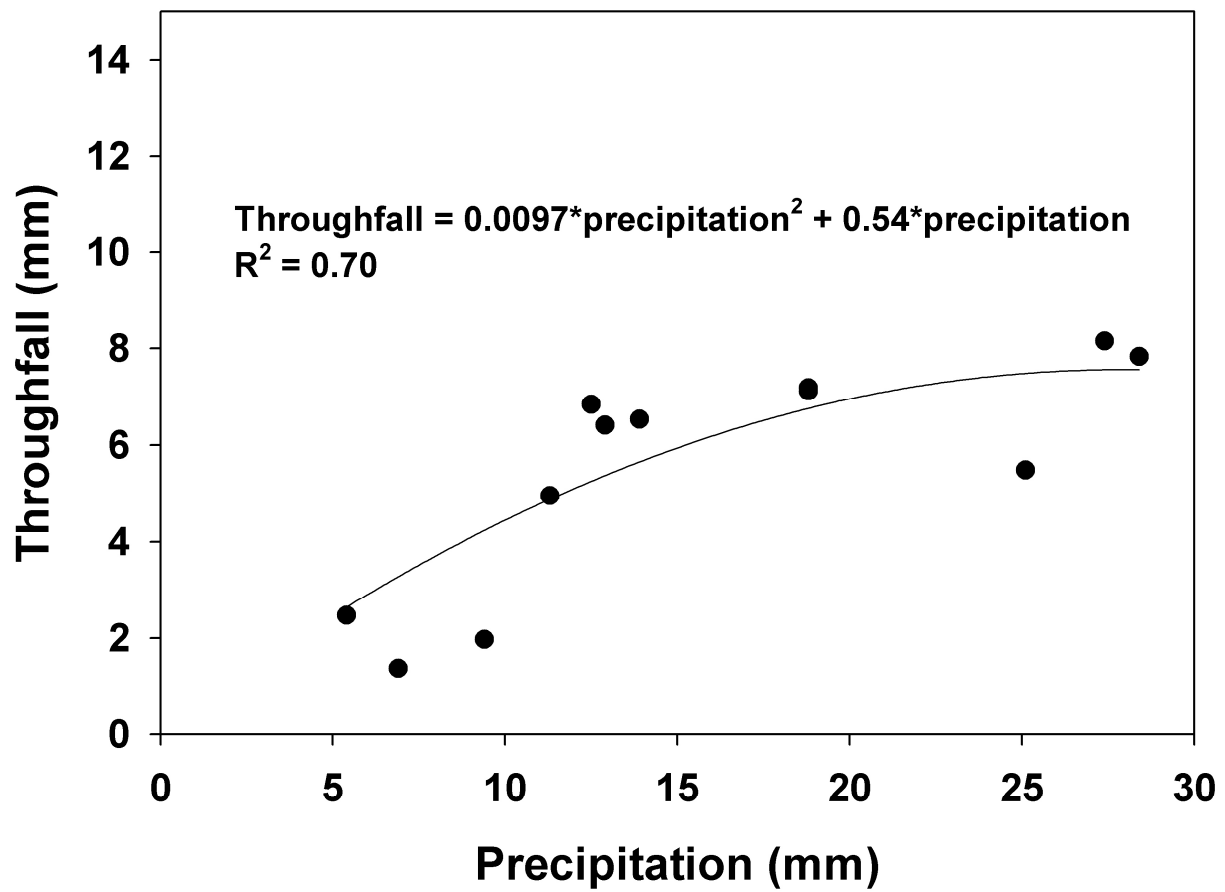


Figure 4-5. The comparison of throughfall and precipitation at the woodland site.

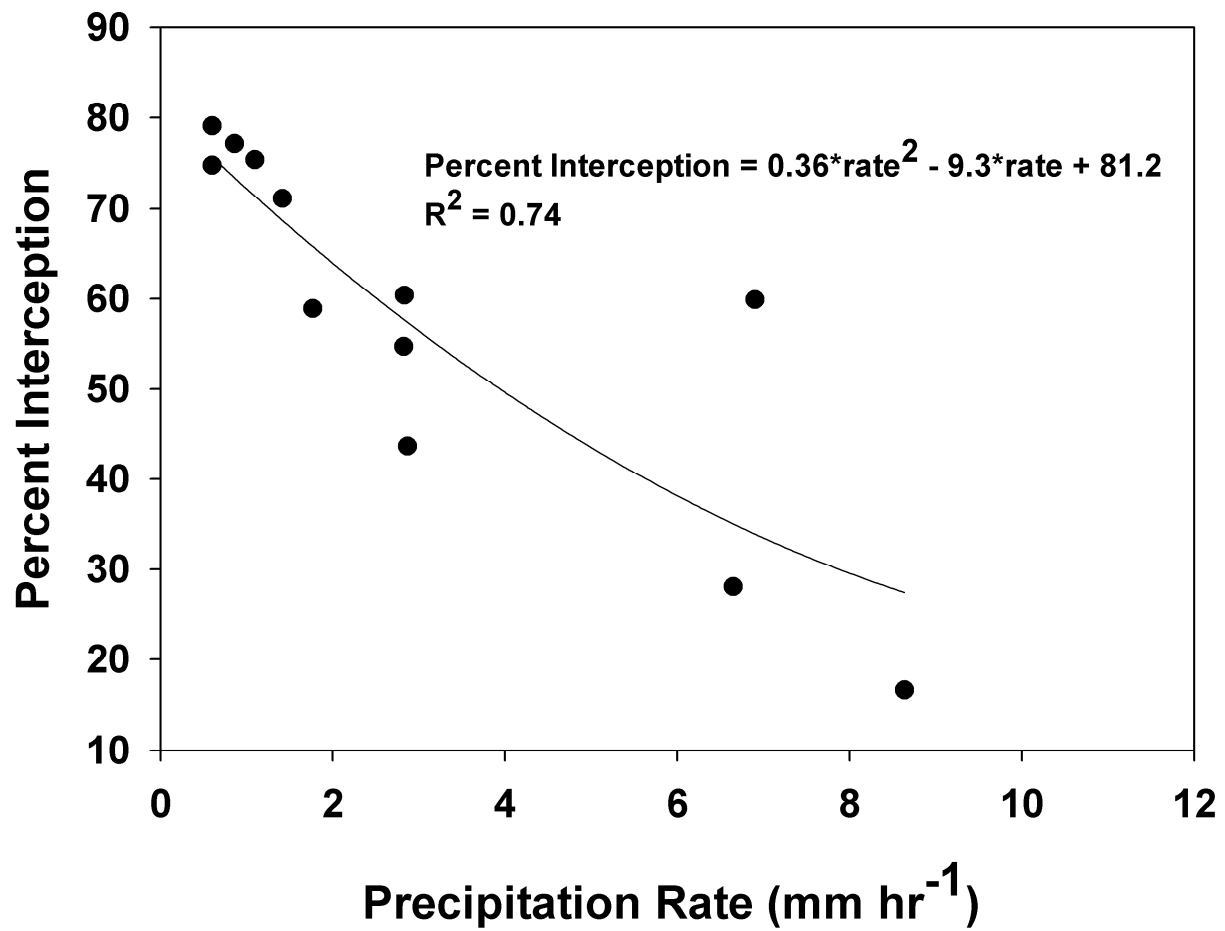


Figure 4-6. Percent interception compared to precipitation rate at the woodland site.

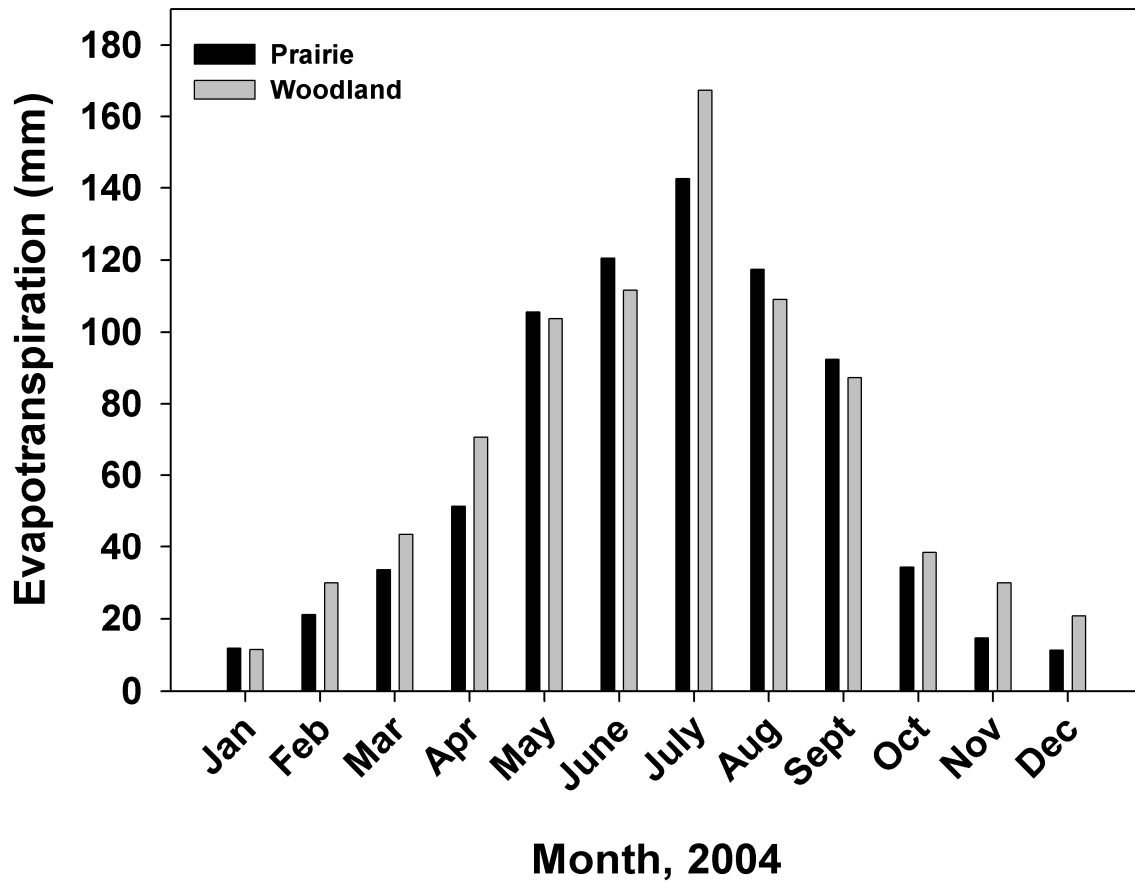


Figure 4-7. The monthly evapotranspiration for the Prairie and Woodland sites in 2004.

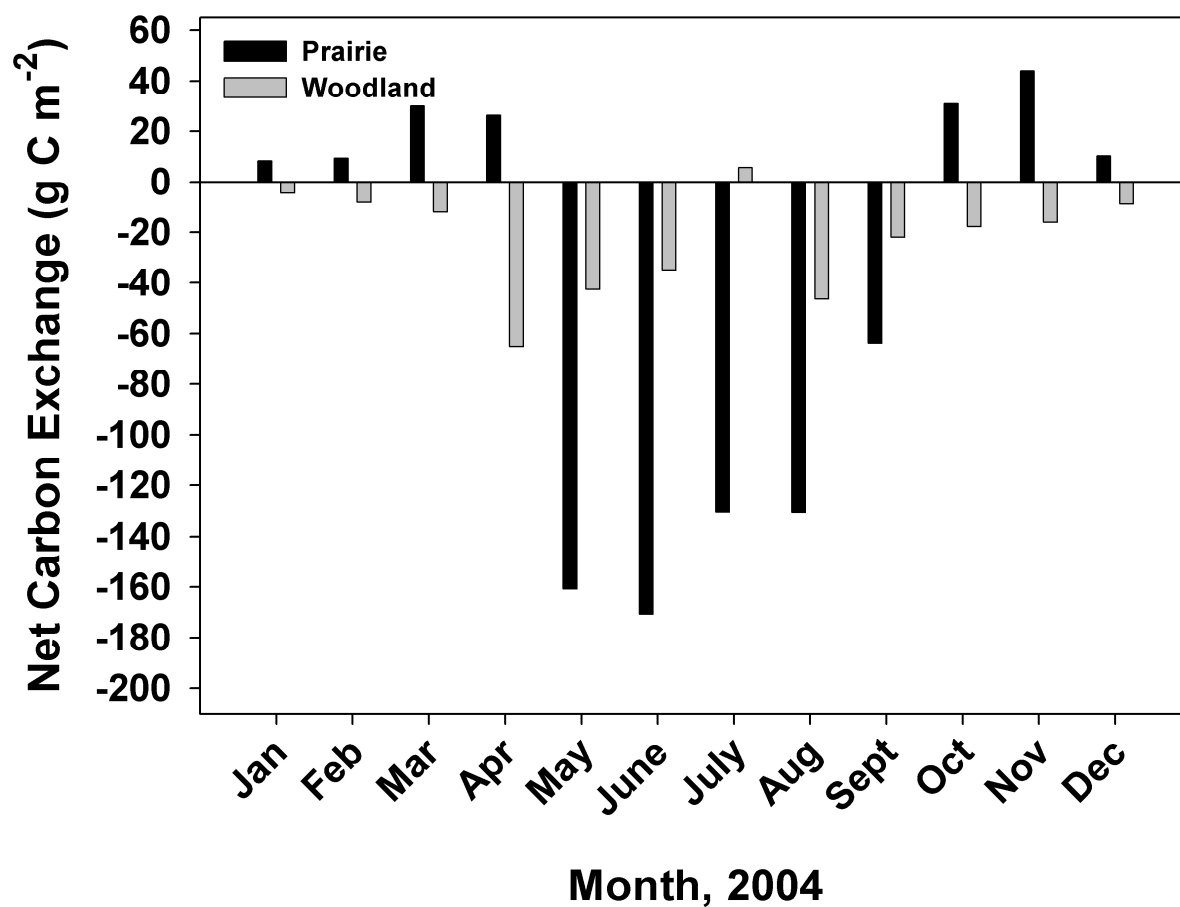


Figure 4-8. The monthly NCE from the Prairie and Woodland sites in 2004. The negative values represent a gain of carbon to the system (i.e., net downward flux).

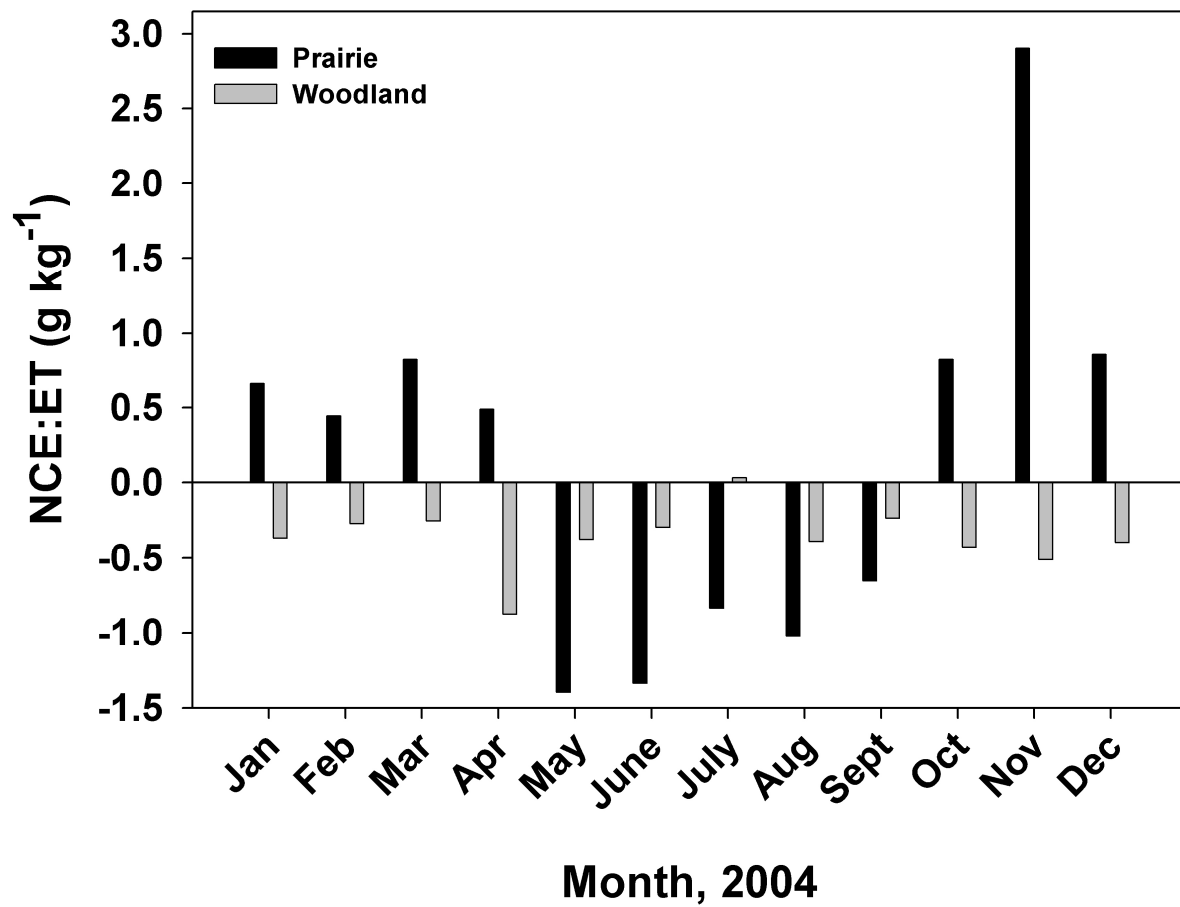


Figure 4-9. The ratio of NCE to ET compared between the prairie and woodland sites.

Negative numbers represent a gain of carbon to the system.

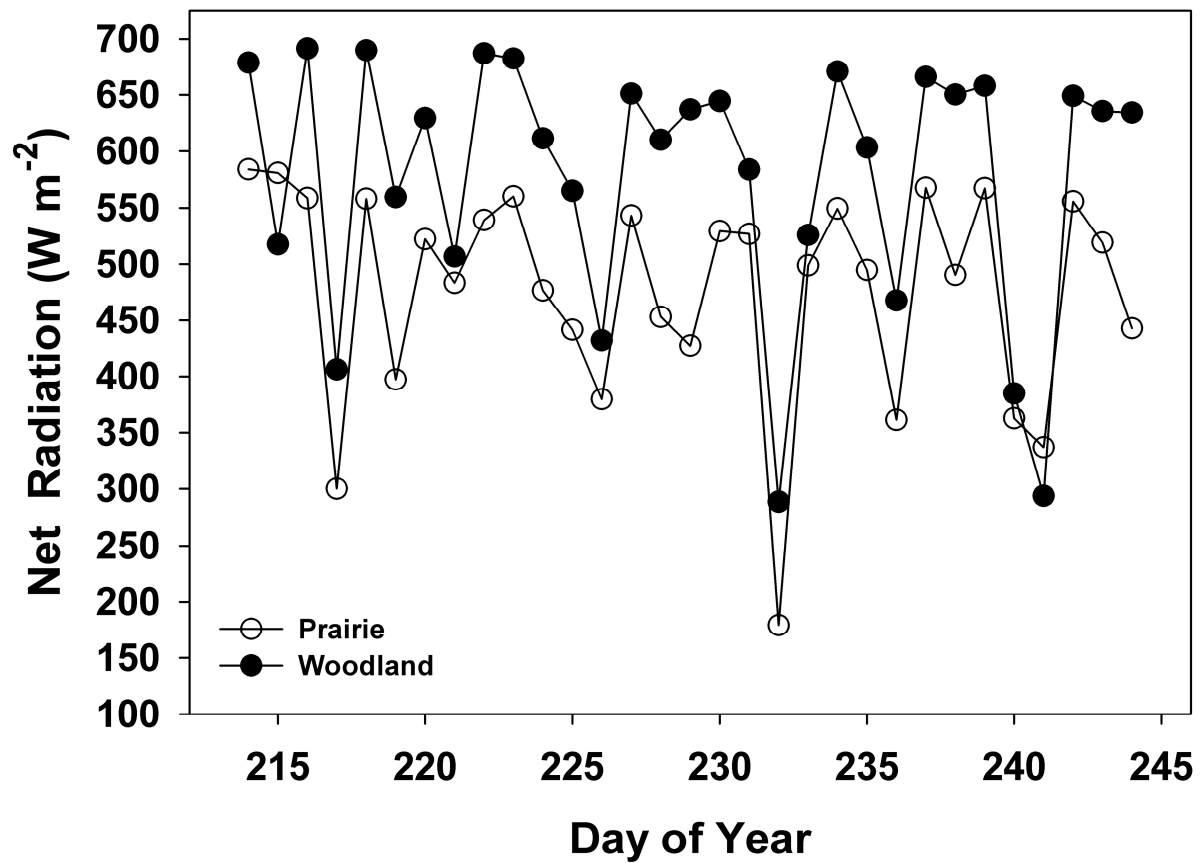


Figure 4-10. The midday net radiation compared between the prairie and woodland sites during August, 2004. Data are averages calculated from 11:00 to 15:00 h local standard time.

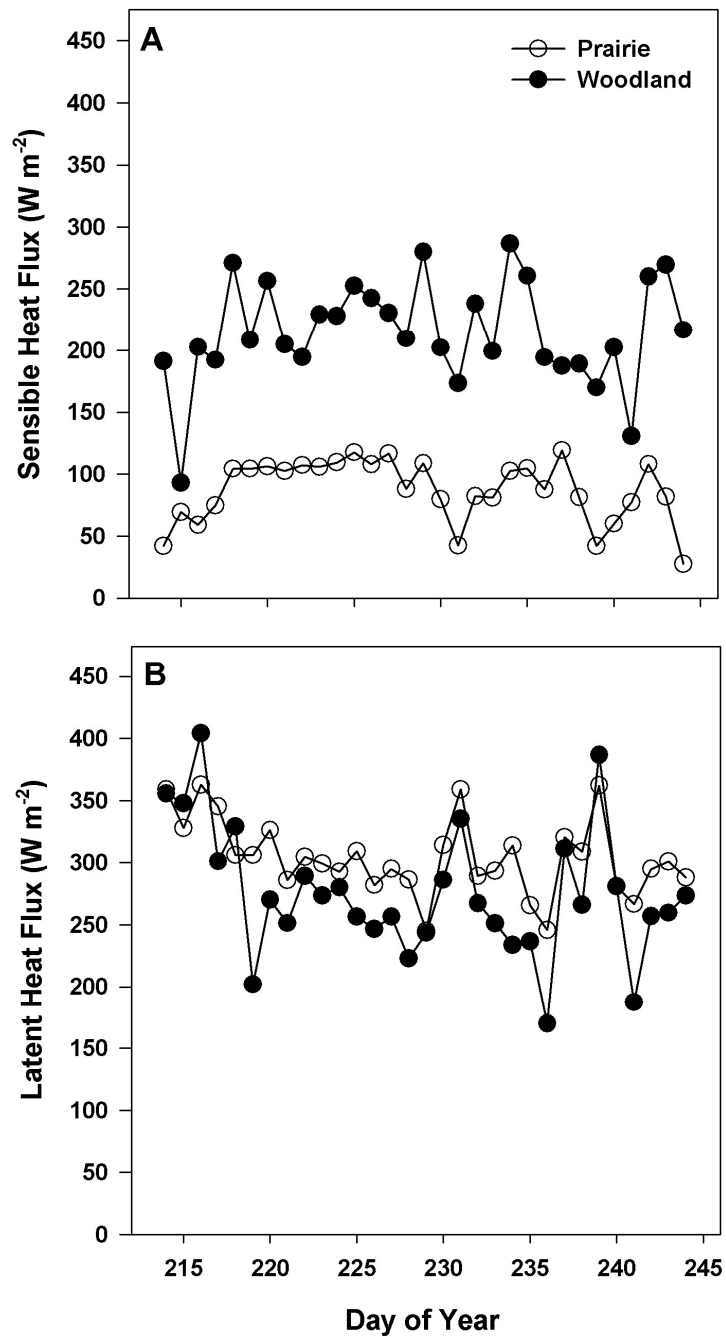


Figure 4-11. Midday sensible heat flux (A), and latent heat flux (B), compared between the prairie and woodland sites during August, 2004. Data are averages calculated from 11:00 to 15:00 h local standard time.

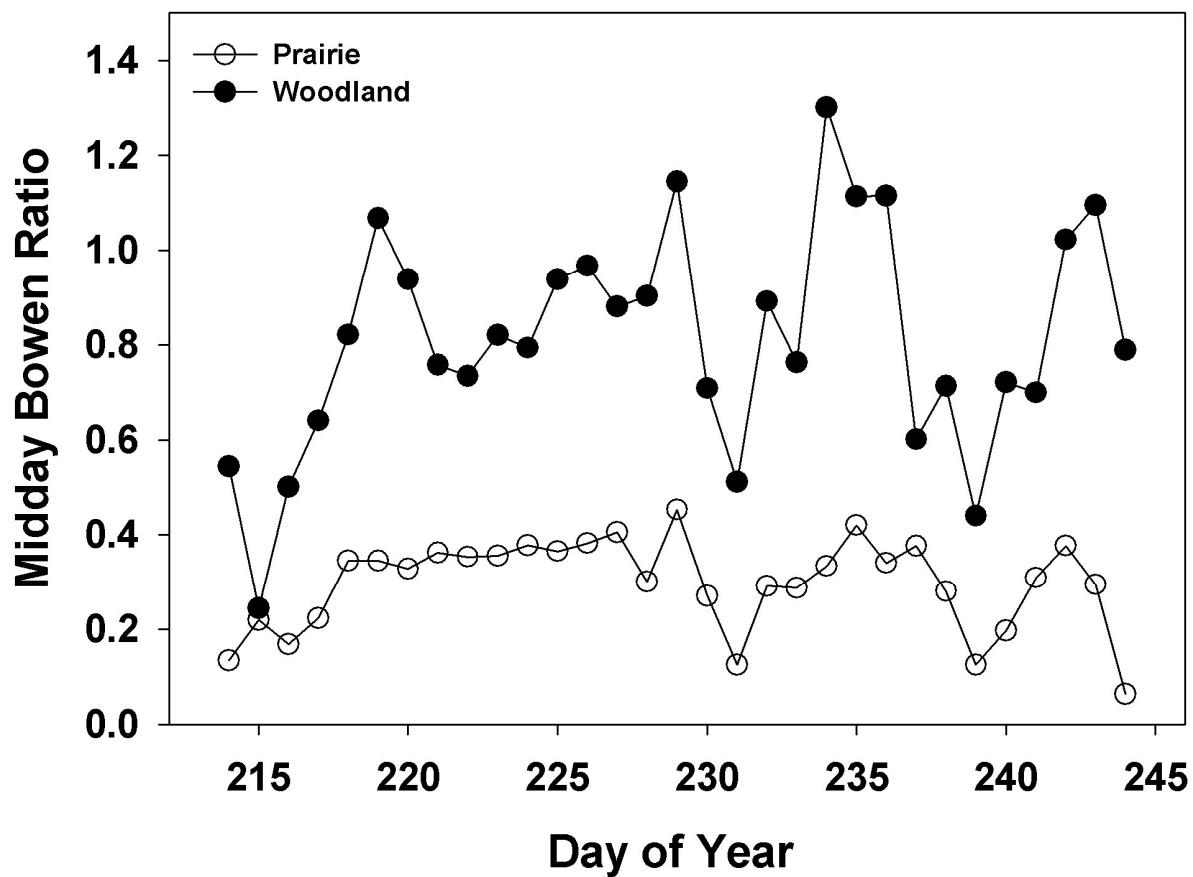


Figure 4-12. Midday Bowen ratio (H/LE) compared between the prairie and woodland sites during August, 2004. Data are averages calculated from 11:00 to 15:00 h local standard time.

Table 4-1. Recorded throughfall by individual sample location. Also included is the ratio of throughfall and precipitation.

Date	Precipitation	Location 1	Location 2	Location 3	Mean Throughfall*	Throughfall : Precipitation
	mm	mm	mm	mm	mm	mm/mm
10/27/2004	5.4	2.7	2.4	2.3	2.5 +/- 0.2	0.5
11/3/2004	11.3	5.0	4.7	5.1	5.0 +/- 0.2	0.4
11/12/2004	13.9	7.0	6.0	6.7	6.5 +/- 0.5	0.5
11/19/2004	9.4	1.9	1.8	2.2	2.0 +/- 0.2	0.2
4/13/2005	28.4	8.1	7.3	8.0	7.8 +/- 0.4	0.3
4/28/2005	18.8	6.0	9.9	5.5	7.1 +/- 2.4	0.4
5/16/2005	25.1	4.6	6.5	5.3	5.5 +/- 1.0	0.2
6/1/2005	27.4	8.5	7.7	8.2	8.2 +/- 0.4	0.3
6/22/2005	6.9	1.4	1.4	1.4	1.4 +/- 0.0	0.2
7/6/2005	18.8	7.1	6.9	7.6	7.2 +/- 0.3	0.4
7/20/2005	12.5	6.6	6.5	7.4	6.8 +/- 0.5	0.5
8/26/2005	12.9	5.3	6.8	7.1	6.4 +/- 1.0	0.5
Average	15.9	5.4	5.7	5.6	5.5 +/- 0.6	0.4

* mean \pm one standard deviation.

Table 4-2. Recorded stemflow by individual collector.

Date	Tree 1	Tree 2	Tree 3	Tree 4	Tree 5	Mean Stemflow*
	(L)	(L)	(L)	(L)	(L)	(L)
10/27/2004	0.1	0.5	—	—	—	0.3 +/- 0.2
11/3/2004	0.2	0.9	—	—	—	0.6 +/-0.5
11/12/2004	0.3	1.1	—	—	—	0.7 +/- 0.5
11/19/2004	0.4	—	—	0.5	0.4	0.4 +/- 0.1
4/13/2005	0.1	0.1	0.0	0.5	0.1	0.1 +/- 0.2
4/28/2005	0.1	0.2	0.1	0.0	0.4	0.2 +/- 0.2
5/16/2005	3.4	3.1	4.5	—	5.3	4.1 +/- 1.0
6/1/2005	7.2	6.7	—	—	7.8	7.2 +/- 0.5
6/22/2005	2.0	2.0	4.3	4.7	3.2	3.3 +/- 1.2
7/6/2005	8.0	7.9	7.8	8.0	7.8	7.9 +/- 0.1
7/20/2005	8.3	8.6	8.2	8.5	8.1	8.3 +/- 0.2
8/26/2005	5.2	5.4	4.2	8.0	10.5	6.7 +/- 2.6
Average	2.9	3.3	4.2	4.3	4.9	3.3 +/- 0.6

* mean \pm one standard deviation.

Table 4-3. Annual evapotranspiration (ET), net carbon exchange (NCE), and the ratio of NCE to ET compared between the woodland and prairie sites. Negative numbers represent a sink (or gain) of carbon to the system.

Location	ET (mm)	NCE (g C m⁻²)	NCE:ET (g kg⁻¹)*
Woodland	885	-264	-0.3
Prairie	813	-506 (-288)**	-0.62

* a mm of evaporation is equal to a kg of water per m².

** includes carbon lost in the burn.

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CHAPTER 5 - Conclusion

These micrometeorology studies investigated how changes land management practices such as spring burning and grazing may impact the evapotranspiration (ET) and water balance of the tall grass prairie in the Flint Hills of eastern Kansas.

Results of the stock pond study show which components of the water balance are most important and may provide background information for an improved design framework for ponds on rangeland. On average, evaporation accounted for 64% of the water loss, followed by seepage of 31%, cattle use of 3% and transpiration of 2%. Since water losses were high from evaporation and seepage, construction of ponds that are deeper and have less surface area, as well as compacted soil liners may improve water conservation of stored water. This research demonstrates that adequate technology and knowledge is available to provide site-specific designs for stock watering ponds and livestock watering strategies in the Great Plains. In areas where livestock drinking water often becomes limiting, having properly designed and managed stock watering ponds could have significant economic benefits.

Comparisons of grazed and ungrazed areas showed that grazing caused only small, ~4 % reductions in ET compared with ungrazed pastures despite large differences in vegetative cover. The largest difference in ET was observed in 2006, when a 6% reduction in grazing was noticed. Biomass and leaf area production was the lower this year than the other years in the study, and is reasoned to have caused the difference in ET. Removal of biomass by grazing causes a regrowth in the grass, resulting in younger leaf tissues being constructed. These results indicate that

removal of biomass does not greatly affect ET, and that changes in ET alone should not affect surface hydrology in grazed areas.

In the woodland study, the 50-yr-old cedar canopy intercepted 54% of the received precipitation, thus decreasing the amount of precipitation reaching the soil. Evapotranspiration from woodland and prairie sites were similar, but net carbon exchange was greater on the prairie. Thus, the apparent water use efficiency during the summer months was 3 times greater on the prairie. Net radiation at the woodland site was 100 W m^{-2} higher compared with the prairie. This caused an increase in the woodland sensible heat flux and midday Bowen ratios, yet woodland latent heat flux and ET was similar to the prairie during the study, factors that could affect regional climate. Cedar encroached areas can intercept over half of the received precipitation, which may have impacts on infiltration and surface water flows. As cedar forestation increases, less water from precipitation will reach the soil surface. This will lead to diminished surface water flows from forested areas. As surface water flows decrease, the amount of water that is channeled into stock watering ponds and reservoirs will decrease, causing shortages in available drinking water for grazing livestock.

These experiments were designed to estimate or measure the water and energy balance parameters, and ET between grazed and ungrazed prairies. Unfortunately these studies only used one pond, one woodland location, and one grazing treatment. Therefore it is uncertain from these studies alone just how lack of management will impact the tallgrass prairie. More experiments are needed to explore several ponds, different grazing strategies, and woodland areas, with in several years to gain a better perspective on how management practices can impact the tallgrass prairie. Management decisions disregarding grazing, prescribed burning, and stock pond design will impact the watershed hydrology and productivity of the tallgrass prairie.